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Epidemiologic Investigation to Identify Chronic Health Effects of Ambient Air Pollutants in Southern California

CALIFORNIA ENVIRONMENTAL PROTECTION AGENCY
AIR RESOURCES BOARD
Research Division

EPIDEMIOLOGIC INVESTIGATION TO IDENTIFY CHRONIC HEALTH EFFECTS OF AMBIENT AIR POLLUTANTS IN SOUTHERN CALIFORNIA

Phase II Final Report
Contract No. A033-186

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EXECUTIVE SUMMARY

1. Background, Aims and Approach

Air pollution in Southern California continues to pose significant challenges to regulatory agencies and to health professionals. Over 12 million people currently reside in the region, and many population projections predict that this number will continue to increase. Several million persons living in the region are exposed to pollution levels that have been associated, in laboratory and field investigations, with acute and subacute respiratory effects. Reported laboratory observations include decrements in pulmonary function, increased prevalence of respiratory symptoms, and respiratory tract inflammation. The paramount question is whether chronic respiratory disease occurs as a result of breathing this polluted air.

To address this question we have embarked on a three-phase research project. The first phase (Phase I) was to evaluate and consider the best approaches to follow in measuring exposures, in assessing health effects and in selecting populations for study. This phase was completed and reported on in 1992 (Peters, et al., 1992). The second phase (Phase II), the actual cross-sectional study of children living in Southern California, is the subject of this report. Children from 12 communities were studied during the school year ending in 1993. This was repeated during the school year ending in 1994. Phase III is underway and is following the children identified in 1993 augmented by about 2000 new fourth grade children from the school year 1995-96.

The Phase II cross-sectional study was conducted to provide early information on the possible chronic effects of air pollution in Southern California children and to determine, if effects are found, which pollutant (or pollutants) is responsible. To do this, about 3600 children from 12 communities with differing air pollution patterns were studied. About one-half of the children were in the fourth grade, about one-quarter in the seventh and about one-quarter in the tenth.

Methods

Three approaches were followed to assess health effects and to collect other needed information. These were questionnaires, pulmonary function testing and school absence monitoring to determine frequency and severity of health problems. Annual questionnaires were completed on these children which covered health history (including history of wheezing, asthma, bronchitis, pneumonia and other respiratory conditions), residential history, housing characteristics (such as heating and air conditioning practices), and history of exposure to other possibly harmful agents, such as tobacco smoke (both active and passive smoking). In addition, the usual physical and outdoor/indoor activity of each subject was ascertained.

The lung function of each subject was assessed annually to determine ventilatory capacity. School absences were recorded to determine frequency and severity of respiratory illnesses. The results of these health outcomes were correlated with air pollution measurements to determine if an association was present.

The communities were selected on the basis of extremes of exposure and by profile of pollutants. Some of the twelve communities did not have monitoring stations and existing stations needed enhancements to achieve accurate estimates of the exposures of health interest. Since the study was designed to allow for chronic effects we needed to develop special instruments that could provide information on potentially important exposures for which little or no previous information was available. This resulted in the production of the two-week sampler that collected data on $PM_{2.5}$ and acid vapor.

After the development and deployment of the instrumentation, monitoring for air pollutants was conducted for the twelve communities, the schools and a sample of the subject's residences. Ozone, PM_{10}, $PM_{2.5}$, NO_2, and acid vapor concentrations were determined at the community level, and indoor ozone concentrations were measured at schools. A sample of homes was measured for indoor ozone, PM_{10}, $PM_{2.5}$, acid and formaldehyde. The information from the questionnaire on residential history allowed for the construction of an estimated life-time exposure level for the different pollutants based on existing data. The information collected at schools and homes allowed for adjustments for exposures based on whether the subjects were indoors or outdoors.

The study design allows for community comparisons, individual comparisons and a combined analysis. This is important because individuals may not have spent their entire lives in one community allowing for potential migration from low pollution to high pollution or vice versa. This report describes the progress to date through the end of Phase II which was completed in May 1995. Analyses are continuing.

2. RESULTS

Exposure Assessment

Two-week aerosol/acid sampler development

An extensive effort involving laboratory component testing, field component testing, and complete system field testing was undertaken to develop a Two-Week Sampler (TWS). The performance of the new sampler exceeded the accuracy requirements for measurements of $PM_{2.5}$ mass, $PM_{2.5}$ sulfate, $PM_{2.5}$ nitrate, and $PM_{2.5}$ ammonium aerosol. For nitric acid, the sampler has a 15- to 30-percent positive bias. Seventeen samplers were built for the project. During the first year of sampling, the samplers had adequate precision (±15 percent) for differentiating the concentrations between the 12 communities.

Timed Exposure Diffusion ozone sampler development

A Timed Exposure Diffusion (TED) ozone sampler was developed to facilitate economical and concurrent sampling in a large number of schools. This sampler was designed to enhance the existing passive sampling methods of Petros Koutrakis at Harvard University. Laboratory evaluation of the sampler showed that it was not affected by interference from NO_2, nitrous acid, PAN, or SO_2. However, it does respond to nitric acid and hydrogen peroxide, which both generally have low ambient and indoor concentrations relative to ozone. Field evaluations showed the device was able to measure ozone with a +6 percent bias and ±12 percent precision on average, which meet the study's requirements. Fifty TED samplers were built for the project and employed during Phase II.

Personal ozone sampler evaluation

The Koutrakis active hollow tube denuder, which uses a sodium nitrite-coated denuder together with a small pump, was tested over a broad range of concentrations in the laboratory chamber. The results showed that it had a lower limit of detection than the passive Koutrakis badge and that it measured ozone with little bias and good precision.

An "all outdoor" experiment was conducted where personal ozone was sampled for 2½ hours on 78 children on two summer afternoons in Riverside. Each child had both the Koutrakis passive ozone monitor and the Koutrakis active hollow tube denuder. Wearing an active denuder in the backpacks did not interfere with the children's activities. The data show the personal ozone levels measured by the active monitoring device were 8 ± 8 percent lower than the stationary continuous monitoring data on average. The passive device ozone data were 18 ±19 percent higher than the stationary continuous monitoring data on average and had outliers. The field evaluation suggested the active hollow tube denuder can provide sufficiently accurate and precise personal ozone data to use in exposure model evaluation studies when the children are adequately supervised. The passive sampling device does not have sufficient precision to meet the needs of the study, although it may be useful for categorizing a subject's ozone exposure as high, medium or low.

Current ambient air quality in the 12 communities

The first full year (December 1993 to December 1994) of measured ambient air quality data in the 12 communities indicates the concentrations of ozone, NO_2, and PM_{10} are generally in agreement with the expected pollutant profiles (see Section 3). Atascadero, Santa Maria, and Lompoc have low concentrations of all pollutants relative to the other communities, and San Dimas, Upland, Mira Loma, and Riverside have moderate or high concentrations for all pollutants. These are good choices for unpolluted and polluted communities. The measured concentrations of gaseous

inorganic acids (HNO_3 and HCl) do not agree as well with the study design profiles as those for ozone, NO_2, and PM_{10}. For purposes of this comparison, low and high concentrations are defined as being more than 0.5 standard deviations below and above the 12 community mean concentrations, respectively. A comparison of the 1994 community air quality rankings with those from the design are shown in Table ES-1.

For ozone, 10 of the 12 communities have the expected profiles based on the annual average daily 1-hr maximum values and all of the communities have the expected profiles based on the May to September "high ozone season" data. The annual average ozone at Upland and Mira Loma was lower than expected, but both communities rank as high for ozone based on the "high ozone season" data. Lake Arrowhead stands out as the highest ozone community.

For NO_2, 10 of the 12 communities have the expected relative concentration rankings. The exceptions are Lancaster, which has moderate NO_2 levels rather than low levels, and Lake Arrowhead, which has low levels rather than moderate levels. These differences are minor and will not affect the ability of the study to detect effects of NO_2. Upland is the highest NO_2 community in the study.

For PM, 10 of the 12 communities have the expected relative concentrations based on PM_{10} and 11 of the 12 communities have the expected relative concentrations based on $PM_{2.5}$. San Dimas and Riverside have moderate PM_{10} levels rather than high PM_{10} levels; however, both have high $PM_{2.5}$ levels. Lancaster has low $PM_{2.5}$ levels rather than the moderate levels expected; however, it has moderate PM_{10} levels. Mira Loma stands out as the highest PM community. The PM_{10} levels at Riverside were surprisingly low compared to those at Mira Loma and the historical PM_{10} levels at Rubidoux; both of these sites are within 15 km of the Riverside monitoring station. Efforts are underway (in Phase III) to investigate the cause of the unexpectedly low PM_{10} concentrations at Riverside.

Prior to our study, very little data for acids were available. Therefore, it was more difficult to classify communities with respect to expected exposure for these pollutants. For inorganic acids (nitric and hydrochloric acids), seven of the 12 communities have concentrations that match the expected profiles (two high-acid, two moderate-acid, and three low-acid communities match). However, Lancaster and Long Beach had low and moderate levels, instead of high levels, and Alpine and Mira Loma had moderate instead of low levels. The largest deviation from the design profile was at Riverside which had high levels instead of expected low levels. The organic acid (formic and acetic acid) levels exceeded the inorganic acid levels at all sites; the relative ranking of communities for organic acids was mostly similar to that for inorganic acids. The absence of low inorganic acid concentrations at Riverside and Mira Loma, where other pollutants are high, may have some effect on the ability of the study to separately detect the effects of acidic species from ozone, NO_2, and PM. The prospective follow-up of

the subjects with interim measures of acid exposure will allow the assessment of whether acid levels influence the growth rate of children's lungs and their school absence frequency and severity.

Overall, the observed ranges of long-term average concentrations between the unpolluted communities and the most polluted communities are in agreement with the design expectations (i.e., factors of three to five differences in annual average concentrations).

Ozone at schools

The TED sampler 8:00 a.m. to 3:00 p.m. integrated ozone data indicate indoor ozone levels were low relative to outdoor levels in almost all schools. The indoor concentrations ranged from below the detection limit (8 to 10 ppb) to 40 ppb, while the outdoor concentrations ranged from 10 to 120 ppb. Almost half of the indoor ozone concentrations were below the detection limit of 8 to 10 ppb, which was a somewhat surprising result. The mean indoor/outdoor ratio was 0.32. The highest indoor ozone concentration, 40 ppb, is comparable to the tropospheric background concentration at this latitude. The low indoor ozone levels and low indoor/outdoor ratios are believed to be due to the prevalence of air conditioning in the schools, which usually results in low ventilation rates because the windows are closed when the air conditioning is used.

Within-community variance in ozone

The school outdoor ozone data indicate there was modest within-community variance in ambient ozone concentrations. In most cases, the 8:00 a.m. to 3:00 p.m. integrated ozone concentrations outdoors collected at different schools were within ±20 ppb of the levels recorded at the community monitoring station. In some cases, differences as large as 40 ppb (80 versus 120 ppb) were observed. This would decrease the accuracy of our estimated individual exposures.

Pilot study of children's ozone exposures

A pilot personal ozone monitoring study was conducted on 140 sixth grade students in San Dimas and Mira Loma in October. The ambient ozone concentrations during the study were unusually low and had a small dynamic range (daytime concentrations of 33 to 54 ppb). The pilot study demonstrated the feasibility of using the newly developed active personal ozone sampler on children for short-term (4 to 9 hours) sampling along with hourly time-activity diaries. For part of the sampling, the children were unsupervised and were required to assemble and disassemble the sampler. The active samplers had adequately low limits of detection for the experiment, but its

precision was substantially poorer than in the summer field evaluation in Riverside. Subsequent to the pilot study, the HSPH researchers re-engineered the active sampler to address the leakage and assembly/disassembly problems discovered in this pilot study.

The personal ozone data showed there was high variability among the students' exposures and the average personal ozone levels were only 28 percent of the ambient. Analysis of the individual data indicated that the ambient ozone concentrations explained only 6 percent of the variance in personal exposure. However, when the personal ozone data were grouped by sample type and school, the ambient ozone explained 46 percent of the variance in average personal ozone. This result clearly establishes the linkage between personal and ambient ozone.

An exposure model that used the ambient concentrations in conjunction with the students' time-activity diaries explained 21 percent of the variance in personal ozone exposure. The low levels and lack of dynamic range in the ambient ozone concentrations along with the poorer precision of the data limited the utility of the data for model evaluation. Future personal ozone experiments should be conducted in summer, rather than October in order to capture a broader range of concentrations, and with re-engineered personal samplers to improve precision.

Since children spend a significant amount of time at their homes, characterization of air quality in and near residences was considered essential. During Phase II, the ARB sponsored an independent residential air monitoring program to address this need. Measurements were made at 121 homes in four high-pollution communities, that included air exchange rates, and 24-hr concentrations of ozone, $PM_{2.5}$ mass, and formaldehyde indoors and outdoors. On average, indoor ozone levels were less than 40% of outdoor levels. Average PM_{10} and $PM_{2.5}$ levels were slightly higher indoors while the maximum levels were higher outdoors. Formaldehyde concentrations while generally low, were higher indoors as expected.

Table ES-1. Ranking of the mean ambient air quality conditions in the 12 communities in the study design and in 1994.

Community	Ozone Ranking		NO_2 Ranking		PM_{10} Ranking		$PM_{2.5}$ Ranking	Inorganic Acids	
	Study Design	May-Sept 1994	Study Design	Annual 1994	Study Design	Annual 1994	Annual 1994	Study Design	Annual 1994
Atascadero	low	low	low	low	low	low	low	low	low
Santa Maria	low	low	low	low	low	low	low	low	low
Lompoc	low	low	low	low	low	low	low	low	low
Lancaster	mod[a]	mod	low	mod	mod	mod	low	high	mod
San Dimas	high	high	high	high	high	mod	high	high	high
Upland	high	high	high	high	high	high	high	high	high
Long Beach	low	low	high	high	mod	mod	mod	high	mod
Mira Loma	high	high	high	high	high	high	high	low	high
Riverside	high	high	high	high	high	mod	high	low	mod
Lake Elsinore	mod	mod	mod	mod	mod	mod	mod	mod	mod
Lake Arrowhead	high	high	mod	low	low	low	mod	mod	mod
Alpine	mod	mod	low	low	low	low	low	low	mod

a low ≤ 0.5 standard deviations below the 12 community mean.
mod (= moderate) is between -0.5 and +0.5 standard deviations of the 12 community mean.
high ≥ 0.5 standard deviations above the 12 community mean.

Descriptive Statistics and Results of Health Effects Analyses

Population

3701 children approximately evenly distributed among 12 communities were evaluated by questionnaire during the school year in 1993. 3248 children performed pulmonary function tests in the Spring of 1993. About one-half of these children were in the 4th grade, one quarter in the 7th grade and one-quarter in the 10th grade. The ages ranged from 8 to 17 years old. About 89% of these subjects were restudied with pulmonary function tests during the Spring of 1994.

Demographics

The age distribution was similar among the communities. In 9 of 12 communities, whites were in the majority. Santa Maria was the only predominantly Hispanic community. Most subjects came from families of middle income or higher. Lancaster and Santa Maria had slightly lower incomes. Most students were born in California and of those born outside of California, most were born in the U.S.

Residential Characteristics

Most subjects live in single family houses. Long Beach had older homes than other communities. The average number of bedrooms is between three and four in every community but Long Beach and Santa Maria where the number is lower. Gas stoves predominate in every community but Alpine. The vast majority of homes have microwave ovens, and carpet in the child's bedroom. Air conditioning use varies considerably by community as was expected given differences in climate between cooler coastal and hotter inland communities.

Health Effects by Community Comparisons

The three health outcomes; pulmonary function level, symptom prevalence, and school absence caused by illness, were evaluated by looking for community differences. Pulmonary function levels did not vary significantly between communities. Respiratory disease prevalence as assessed by questionnaire did vary between communities but was not higher in communities with high air pollution levels. Illness absence rates varied between communities but was not higher in communities with high air pollution levels. These three findings suggest that no profound health effects are occurring as a result of air pollution.

When individuals within in each community were assigned that community's historical air pollution levels, some statistically significant results were seen with adverse, subtle changes in pulmonary function occurring as air pollution levels increased. Statistical analysis could not identify a specific pollutant responsible for

these associations. No significant associations were seen between respiratory disease prevalence and historical air pollution exposures.

Health Effects by Individually Assigned Lifetime Exposure to Air Pollutants

Pulmonary Function

When each individual's lifetime exposure to air pollution was analyzed with respect to the health data, statistically significant associations were seen with ozone being strongly associated with lowering of pulmonary function especially in male subjects. The expiratory flow rates were the most consistently affected. In boys, the estimated effects of lifetime average O_3 exposure on FEV_1 was - 3.22 ml/ppb (95% Confidence Interval (CI): - 5.47, -0.96) and for average peak O_3 was -6.43 ml/ppb (95% CI: -10.89, -1.96).

There was also a tendency for the effects to be larger in the older children suggesting a cumulative effect of exposure. Adjustment for outdoor and physical activity produced a slight increase in the apparent effect of air pollution on pulmonary function. When only the 1994 air pollution measurements were used to calculate exposure estimates, no associations were seen between air pollution level and effects on pulmonary function suggesting that either long-term or past exposure is more important.

Disease Prevalence

Bronchitis rates determined by questionnaire were associated with lifetime exposure to PM_{10} (OR = 1.03, 95% CI 1.00 - 1.05 per ug/m^3). The rates were higher in the female subjects. Asthma prevalence determined by questionnaire was not significantly associated with pollution level.

School Absences

Asthmatics were more likely to be ill than non-asthmatics and to have a higher proportion of respiratory illnesses but recent air pollution levels (previous 1 or 2 days) did not correlate with absence rates in either asthmatics or non-asthmatics. Lifetime exposure to peak O_3 was associated with higher rates of respiratory illnesses and both peak and average ozone were associated with greater duration of absences from respiratory illnesses for all subjects (asthmatics and non-asthmatics).

Conclusions on Health Effects

On the basis of our Phase II results we conclude that long-term exposure to O_3 is associated with subtle adverse changes in pulmonary function and higher rates and duration of respiratory illnesses as assessed by school absence monitoring. Long-term exposure to PM_{10} is associated with a greater prevalence of bronchitis as assessed by questionnaire.

The Phase III follow-up of these children is underway to clarify the causality of these relationships and to determine permanency. Our Phase II findings have provided several important issues to be pursued. For example, we found that boys spend more time outdoors, and more time engaged in strenuous physical activity, than girls do. The difference in the magnitude of the ozone effect between boys and girls, and its potential relationship with differences in time activity patterns will be addressed more precisely with models that adjust exposure measurements by taking time outdoors and physical activity into account.

The findings that adverse effects of exposure tend to be greater in older children is suggestive of a cumulative effect. Longitudinal methods will enable us to estimate the magnitude of this effect as a function of time. This will provide us with a description of changes in lung growth which may occur in response to air pollution exposure.

1. INTRODUCTION

Air pollution in Southern California continues to pose significant challenges to regulatory agencies and to health professionals. Over 12 million people currently reside in the region, and many population projections predict that this number will continue to increase. Several million persons living in the region are exposed to pollution levels that have been associated, in laboratory and field investigations, with acute and subacute respiratory effects. Reported laboratory observations include decrements in pulmonary function, increased prevalence of respiratory symptoms, and respiratory tract inflammation. The paramount question is whether chronic respiratory disease occurs as a result of breathing this polluted air.

To address this question we have embarked on a three-phase research project. The first phase (Phase I) was to evaluate and consider the best approaches to follow in measuring exposures, in assessing health effects and in selecting populations for study. This phase was completed and reported on in 1992 (Peters, et al., 1992). The second phase (Phase II), the actual cross-sectional study of children living in Southern California, is the subject of this report. Children from 12 communities were studied during the school year ending in 1993. This was repeated during the school year ending in 1994. Phase III is underway and is following the children identified in 1993 augmented by about 2000 new fourth grade children from the school year 1995-96.

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Annual questionnaires were completed which covered health history, residential history, housing characteristics, and history of exposure to other possibly harmful agents, such as tobacco smoke (both active and passive smoking). In addition, the physical and outdoor/indoor activity of each subject was ascertained. The lung function of each subject was assessed annually and school absences were recorded to determine frequency and severity of respiratory illnesses.

Monitoring for air pollutants was conducted for the communities, the schools and a sample of the subjects' residences. Ozone, PM_{10}, $PM_{2.5}$, NO_2, and acid concentrations were determined at the community level. Ozone, respirable particulates, formaldehyde, air exchange rates and airborne acids were measured at a sample of residences. Ozone was measured inside and outside at the schools. This report describes the progress and results of the cross-sectional study, completed in May 1995.

1.1 **Aims** - The aims of this project, which consists of a series of investigations, are to determine whether air pollution in Southern California causes chronic respiratory disease or changes in lung function development in humans, and if so, to determine whether specific pollutants are responsible for these effects. It is also likely that if chronic respiratory effects are identified, that these investigations will yield information useful for the setting of ambient air quality standards to prevent these chronic lung conditions.

1.2 **Rationale for the Investigations** - The rationale for this study is based on the premise that identifying environmental causes of human illness requires attention to 3 major areas: 1) studying the most sensitive and objective end-points of health effects, 2) assessing relevant environmental exposures accurately and precisely and 3) using the most powerful study design and statistical procedures. Phase II was built on these cornerstones. A complete description of the deliberations that went into refining the study are presented in two reports, the Phase I Final Report and Phase II Protocol (Peters, et al., 1992a) and the Addendum to the Phase I Final Report and Phase II Protocol (Peters, et al., 1992b).

Children were selected for study for several reasons: they spend more time outdoors; they exercise more than adults; they do not smoke (at least the young ones); they do not have hazardous occupations; they are more likely to have spent their entire lives in Southern California; their growing lungs may be more sensitive to the effects of air pollution and they are accessible in large numbers through schools.

1.3 **Background** - In short-term exposure studies of humans in controlled exposure chambers, among common air pollutants, ozone shows the strongest evidence of adverse effects. Numerous laboratory exposure-response studies of human volunteers have shown that lung function losses, respiratory irritant symptoms, and increases in bronchial reactivity result from ozone exposures levels attained in the South Coast Air Basin (SoCAB), either from comparatively brief (≈1 hour) exercise at "alert" concentrations of 0.2 ppm and higher or from prolonged exercise at concentrations near the federal primary ambient air quality standard of 0.12 ppm [EPA, 1986; Folinsbee et al., 1988; Lippmann, 1989, 1991a]. Recovery to normal function levels typically takes several hours after ozone exposure ceases. Some effects of short-term exposure persist for more than 24-hours. Similar acute effects have not been seen from other pollutants at the levels encountered in Southern California (NOx, PM_{10} or acid vapors).

Studies of humans in Southern California suggest the possibility of chronic respiratory effects from air pollution (Detels et al., 1987, Abbey et al., 1991; Sherwin, 1991), but because of population attrition in the Detels studies, absence of objective data in the Abbey study, and possible confounding in the Sherwin study, conclusions are uncertain. Essentially no human data exist on chronic respiratory effects resulting from specific components of air pollution. The large number of persons in Southern California exposed to air pollution, the existing data on acute effects, and the available

air monitoring data provide a unique opportunity to examine chronic health effects resulting from air pollution in humans. Careful construction of lifetime exposures, taking into account temporal and spatial patterns of physical activity; sensitive measures of pulmonary responses; and thoughtful consideration of study design, confounding, and bias provide the framework for answering important questions raised by our current knowledge.

One recent cost-benefit analysis of the financial and health-associated costs of controlling ozone concluded that the costs exceeded the benefit (Krupnick and Portney, 1991). The benefit analysis was related to mitigation of acute effects because "...only acute health effects have been linked convincingly to ozone concentrations." This statement has two possible explanations: either urban ozone does not produce chronic effects, or the appropriate studies have not been done. Should chronic effects result from urban ozone exposure, the cost-benefit balance could be significantly shifted (Hall et al., 1992). It is the investigative team's and ARB staff's collective belief that the key studies needed to answer the important public health questions about chronic effects of air pollution have not yet been completed.

1.4 Hypotheses to be Tested

The project seeks to address four critical hypotheses:

(1) Children exposed to levels of air pollution existing in Southern California suffer deleterious chronic pulmonary effects (as measured by effects on pulmonary function), compared with children in low pollution areas.

(2) Children exposed to higher levels of air pollution suffer greater rates of acute respiratory illness, more severe respiratory illness, and exacerbation of underlying respiratory disease (viz more asthma attacks if they are asthmatics), compared with children in lower pollution areas.

(3) There is a subpopulation of children in Southern California who are more susceptible to the effects of air pollutants than others.

(4) The effects of vapor-phase strong acids (HNO_3 and HCl) and organic acids can be differentiated from other pollutant effects.

2. OVERVIEW OF STUDY DESIGN

2.1 INTRODUCTION

This section describes the overall approach to carrying out the Phase II Cross-sectional study. It consists of a section on community and school selection and student enrollment followed by three sections following the divisions outlined above: 1) exposure assessment, 2) health effects assessment, and 3) study design and statistical approaches.

2.2 COMMUNITY, SCHOOL, STUDENT SELECTION

2.2.1 We contacted potential schools to seek participation in the testing program. General selection criteria were location in a preselected community of interest based on air pollution levels and patterns; sufficient population of target-aged children; preponderance of children attending school from the immediate neighborhood; demographic similarity with other potential and participating community school sites; absence of localized air pollution sources and proximal location to a fixed-site air monitoring station.

The design approach specified child entry into the study at the fourth, seventh, and tenth grade (nominally in the following age groups - 9-10, 12-13, and 15-16, respectively) and required the enrollment of at least four schools in each community (two elementary schools, a junior high school, and a senior high school). The community population studied consisted of about 150 fourth-graders, 75 seventh-graders, and 75 10th-graders.

We contacted the appropriate school district superintendents to establish cooperative understandings. When appropriate, we met with the district personnel to present an overview of the study, including the rationale, the goals, and the envisioned level of effort required by participating schools and students for project success. Early in the site negotiations process, we reviewed and visited schools being considered for study participation along with district personnel to identify any unforeseen or unappreciated obstacles to inclusion in the study. Such obstacles included local vocational or school-related circumstances not immediately apparent (such as inadequate space for spirometers or a refusal to release children for testing during school hours); other obstacles involved programmatic or logistical issues unacceptable to school administrative or project personnel (such as project staff having access to certain classroom areas or the location of air monitoring instruments in the classroom for extended periods of time). When insurmountable difficulties were identified, alternative sites meeting our specifications were identified.

Following school district approval for participation, we scheduled meetings with the on-site administrators and potentially affected teachers to discuss the project and

obtain support for its performance. Scheduling for administration of the annual questionnaire, lung function testing, and exposure monitoring was done to minimize the disruption of school activities and to facilitate efficient planning of study field operations.

2.2.2 Student Recruitment

Complete classrooms of students were recruited. The cooperation of the student and the granting of informed parental (or legal guardian) consent for participation by minors were required.

Recruitment consisted of presenting a brief overview of the study to potential classrooms. The presentation emphasized the personal health nature of the study and the opportunity to learn about individual health and the lung. To minimize bias, the presentation downplayed the aspect of air pollution. The study procedures and protocols were explained to enable children to gain insight into the level of participation that would be required (completion of questionnaires once per year, lung function testing once per year and absence monitoring with call back of parents).

2.2.3 Student Enrollment

The health history questionnaires were first circulated during January 1993, the enrollment point for subject entry into the study. They were filled out by a combination of parents and children for the older subjects and by parents for the younger subjects. Enrollment was contingent upon receipt of the written informed consent of the child's parent or legal guardian (whose request for signature appears on the first page of the questionnaire) and upon return of the health questionnaire.

Each student was uniquely identified by code number and his/her personal information remained confidential, divulged and identified only by unreferenced subject number for analytical purposes.

2.3 HEALTH EFFECTS ASSESSMENT

2.3.1 The Annual Questionnaire

The project questionnaire is composed of three main sections: a medical history, a housing survey, and a time-activity assessment. Early in Phase II, we pilot-tested the questionnaire among families of USC staff and other project volunteers ineligible for main study participation (e.g., neighbors of staff, community friends). During Phase I, pilot testing of an earlier version was done among volunteers recruited from local pediatricians' offices seeing large numbers of asthma cases.

Two questionnaires were given to participating subjects in the Winter of 1993, the first school year of study. The questionnaire containing the medical history and housing survey was administered in the early winter and the activity section administered in the spring during lung function testing in the schools.

In administering the questionnaire we contacted all project participants in any given community during a single visit and enlisted the active participation of the selected schools. The project field staff arrived on-site to meet with teachers regarding circulation of the survey material prior to its distribution. Questionnaires were precoded by community classroom. Students were instructed (by teachers) to take the questionnaires home that evening, to help their parents or legal guardians accurately complete as much of the survey as possible, and to return the completed and sealed survey to school the following day (the parents and guardians of fourth-graders completed the surveys in their entirety, whereas the seventh and 10th graders usually completed portions of the questionnaires themselves). Spanish translations of the questionnaire were available.

Questionnaires were logged in each day at school by project staff personnel or locally-recruited volunteers. Participants not returning questionnaires were encouraged to complete and return them as soon as possible. Iterative logging and evaluation of the return/completion rate was updated frequently.

Questionnaires were returned to study personnel at USC via personal courier or expedited mailing. As soon as the questionnaires were logged in at USC, we decided either to accept the return rate or to institute an additional distribution. Incentives were offered to classrooms achieving high participation rates. These took the form of games, books or recreational equipment.

Through scheduling and efficient utilization of the project health assessment field staff, virtually all communities simultaneously participated in survey distribution.

The activity assessment portion of the questionnaire was administered during lung function testing field operations in the spring. Subjects were given the time/activity pattern assessment surveys by project testing staff following their lung function testing battery, asked to complete the written surveys, and return the documents to field testing staff within the subsequent 24-hours. Seventh and tenth graders generally completed these at the time of testing in school. Fourth graders were asked to take them home. Due to low 4th grade return rates, large number of 4th grade surveys were completed with parents via telephone interview. During the school year 1993-94, both questionnaires were repeated.

2.3.2 Lung Function Testing

Lung function testing took place in the spring of 1993 and the spring of 1994 to minimize seasonal confounding with intercurrent summer or winter acute air pollution episodes. Spirometers (Morgan Spiroflow) and associated computerized interfaces and data logging capabilities were used to document lung function performance. Testing was performed in a predetermined and administration-approved area of the school (nurse's office, library, multipurpose room). To the extent possible, we performed testing during the morning hours to avoid possible acute effects of potential daily peaks in ambient air pollution.

Project staff tested the subjects individually in a consistent, prearranged manner. A subject initially removed his/her shoes; we next determined standing height and weight (to provide necessary information for standardized nomogram corrections for predictions of lung function performance based on gender, race, height, and weight); then we began testing. In a seated position, pinching their nose shut with their fingers, the subject was asked to perform at least 3 satisfactory maximal expiratory maneuvers. A maximum of 7 efforts were attempted. Prior to performance of successive maneuvers, the subject was privately questioned regarding smoking; this approach has proven more effective than written responses in prior epidemiologic investigations (Speizer, personal communication, 1992). After testing, a subject was given a time/activity assessment survey to complete at school or at home that evening and he/she was instructed to return the completed survey to school that day or the following day (either to project staff or local volunteers).

Up to six testing units (spirometers), operated by trained lung function technicians, were dispatched to conduct field testing in a given community. To facilitate testing, the six units were assigned as needed to fourth-grade, seventh-grade, and 10th-grade testing. In some instances, a local volunteer at the testing location provided assistance in obtaining students for testing and measuring of students' heights and weights.

Students left class in groups of about five or six, at approximately 10-minute intervals, for testing. Staggering the flow of students ensured that technician time was efficiently used and that student time away from classroom activities was minimized.

Testing was scheduled to conform with classroom schedules (Monday through Friday). To accommodate this schedule, field staff traveled on Sunday or Monday, set up equipment and carried out testing on Monday through Friday mornings. Friday afternoons were reserved for data management, administration, and travel.

Each community was visited at least twice (at least one month apart) with half the participating subjects being tested each visit. This approach minimized the potential confounding of intercurrent acute pollution episodes, outbreaks of illness (e.g. flu), or

some other unidentified and uncontrolled variable; it also facilitated completion of testing within one week of on-site presence. The study population was arbitrarily divided (by classroom, or other convenient division) into two roughly equal subgroups, one of which was tested during the initial visit, and the other tested later in the same testing year.

The standard operating procedures and details on pulmonary function testing can be found in Section 5, Report on Quality Assurance and Quality Control. These procedures are described in Linn et al., 1996.

2.3.3 Absence Monitoring

The absence monitoring activity was designed to collect data to determine the frequency and severity of respiratory illnesses in relation to concurrent ambient air pollution levels and to compare respiratory disease patterns between communities and by lifetime exposure to various pollutants.

Because schools are required to keep data on absences to receive capitation funding for students, there is motivation for schools to collect accurate data.

We used the school absence to trigger an investigation to determine the reason for the absence. This involved phoning the student's home to interview the parent or guardian. In deciding what questions to include, literature from studies that used symptom questionnaires in children was reviewed. We decided to let the parent tell us what they thought the illness was but still ask a list of symptoms for each call. The decision was made that the caller could skip the symptom list only if the parent said that the illness was due to a musculoskeletal injury. If the parent said that the absence was for social reasons, we would code that as a non-illness absence and not ask the symptom list. Therefore, we would be able to classify whether the illness was respiratory. We also asked whether the child had seen a doctor and if yes, the doctor's diagnosis was noted. We asked about use of medications since this might provide an indication of the severity of the illness. The decision was made to include comment fields for the interviewers to use because these might provide useful information for future contacts with the parent or guide future modifications of the questionnaire.

In the fall of 1993, a pilot study of absence monitoring was implemented to assess the feasibility of the approach and work out details. In August of 1993, the 6 field technicians began contacting district offices to acquire data on rates of absences.

In their initial contacts with the schools, the field technicians did the following:

1. Ascertained how the districts excused absences: note from home, call from home, or both.
2. Ascertained the method the schools used to record absences; computer

data base versus paper records.

3. Ascertained how frequently the various schools were willing to provide data: weekly, bi-weekly, monthly, quarterly.
4. Attempted to ascertain the coding that schools used for absences -- excused/not excused, ill/not ill.
5. Arranged how the data would be sent to USC - generally by postage paid mailers.

A system to receive data was tentatively arranged in all districts: Out of 52 schools in the study, a system had been set up for 51 by the end of November 1993. Among the schools in which a system was set up, 5 were willing to provide quarterly, twenty eight schools indicated willingness to provide weekly reports, 4 biweekly and 13 monthly. In some schools, particularly middle and high schools, administrators felt that additional staffing was needed to provide these data. For these schools, funds were provided to the schools to pay a part-time clerk 1-2 hours per week to provide these data.

The ease of using the data varied by school. In some schools, students in our study were subsetted so that the absence data included only these children. In most schools data on all children were included and thus the field team was obliged to enter names into our database to ascertain if each child was a study participant. The schools use different codes for the data and different report formats so that it turned out to be more time consuming to input the data than originally anticipated. For each eligible absence, the field technician filled out the top of a calling form with the child's name, name of parent who signed the form, phone number and dates of absence.

Calling sheets were divided among the field personnel. Using a script and structured questionnaire the field technicians made calls to the participant's parents or guardians. Early results on 138 absences, revealed that 31 (22.5%) were reported by parents as not due to illness. Non-respiratory illness accounted for 31 (22.5%) of absences and respiratory illness accounted for 76 (55.0%) of absences.

We also learned that most parents had difficulty remembering absences that were more than 4 weeks in the past, and that 4-8 p.m. was the most efficient time to make telephone contact. Parents were generally willing to provide absence information although some were concerned about how frequently they might get called. The caller often reached another adult in the household who appeared fully knowledgeable about the absence. Because it can be difficult to reach the person who signed the consent form and repeated calls are bothersome to the other household member who is able to provide the information, we decided that after one try to reach the signer, we would accept another adult over the age of 18 as the respondent.

One full-time equivalent of personnel (a project coordinator and two part-time interviewers) dedicated to the absence monitoring activity were hired and trained in January 1994, after the completion of the pilot study. The project coordinator was

responsible for overseeing the receipt and entry of all absence monitoring data, generating a listing of students that needed to be called, distributing them to the two callers, receiving completed surveys and making quality control calls. Although systems had in theory been set up during the pilot activities in the fall, some schools proved resistant to sending the absence data in a timely fashion. In addition, at a number of other schools, even where payment was offered, the project coordinator needed to make frequent contact with the responsible person at the school to receive the reports in a timely fashion. Difficulties getting the district administrator to agree to let us get data directly from the schools in Riverside was a problem that took several months to resolve. Our pilot study had demonstrated that the delay created by having the district collate the data would result in most students having absences more than 4 weeks in the past at the time we received the data. Due to extensive efforts of the project coordinator, a suitable arrangement with the Riverside schools was made. It has turned out that the absence monitoring required much more than one full time equivalent staff person. We have therefore focussed on calling on as many absences as possible that occurred within the 4 weeks prior to receipt of the information from the school. The average number of surveys that can be completed each hour (during the 4 to 8 pm period) by a trained person is 6 for elementary schools and early junior high, and 4 for late junior high and high schools. Parents of high school students are much less likely to be home than parents of younger children, plus truancy problems are more frequent in the older age group. Since we have limited resources to make calls, we have established a priority system which focuses our emphasis on the youngest group.

More detail is provided with regard to absence monitoring in Section 5 and the Quality Assurance and Quality Control document (Appendix C). The results are presented in Section 4.6.

2.4 EXPOSURE ASSESSMENT

2.4.1 Introduction and Background

The objective of the exposure assessment component of this study is to characterize the long-term exposure of the individuals in the study population to ozone, PM_{10}, NO_2 and vapor phase strong acids and organic acids for the prestudy period (birth - 1992), for Phase II (reported here) and for Phase III (the longitudinal study in progress).

Characterization of the nature and extent of actual human exposure to pollutants is critical to understanding the relationships between specific pollutants and human health. Most previous epidemiological studies have used information from central station ambient monitors to characterize the exposures of the study population. These monitors, often the same as those utilized by regulatory agencies for compliance monitoring, are not necessarily located to optimize exposure characterization of the study population. To the extent that true exposures are randomly distributed about the

estimates derived from central site monitors, the capability of the health studies to detect health effects is diminished. The result of random misclassification of exposure in a health study is to bias the results toward finding no health effect. Unfortunately, for many pollutants it is likely that the misclassifications are systematic rather than random. Systematic errors in exposure could lead to either overestimation or underestimation of a health effect.

The actual exposures of individuals to pollutants may largely depend on the nature of the microenvironments in which they live and the amount of time they spend in these various microenvironments. Ambient central site monitors characterize outdoor concentrations at one location in a community. The outdoor concentrations at other locations within the community and the concentrations inside schools, homes, other buildings, and in vehicles are usually substantially different than those observed at the central site monitor. In addition, indoor concentrations are important for characterization of exposure because children spend 85% of their time indoors (Wiley 1991).

The within-community variation (or between-individual variations) in exposure could be particularly important for reactive pollutants like ozone, for pollutants for which there are significant indoor sources like PM and NO_2, and for pollutants that deposit rapidly, like nitric acid vapor. Ozone is formed in ambient air by sunlight acting on NO_2 in the presence of volatile organic compounds (VOCs). Ozone is also scavenged by NO, which is emitted by combustion sources. Compliance air monitoring stations are generally sited so that they are not unduly influenced by local NO sources. However, schools and homes are not; the ozone concentrations outside of homes and schools are probably less than or equal to that recorded by the community monitor. Indoor ozone levels are also less than or equal to ambient levels. Because ozone rapidly deposits on indoor surfaces, it is often far below ambient levels in homes. Indoor PM, NO_2, nitrous acid, and ozone levels may be strongly influenced by combustion emissions from cooking, environmental tobacco smoke, and gas appliances. Within any community, there will be significant variation not only in the types of housing, school buildings, and ventilation habits, but also in human time-activity patterns. Differences in the amount of time spent outdoors and in the subjects' ventilation rates will contribute to the within-community variation in exposures and dosage. It is possible that the within-community variations in exposures could be as large as the between-community variations in ambient levels.

Ambient air pollution concentrations also vary by large amounts hourly, daily, and seasonally as a result of meteorological factors and, to a lesser extent, variations in source strengths and chemical transformation rates. Significant year-to-year differences and long-term trends can also occur. In assessing possible health effects of long-term exposure, one must consider what type or types of exposure are associated with an effect, for example, is an effect associated with frequent acute (short-term, high concentration exposures) or chronic lower level exposures, or a combination of both.

There are two general approaches to exposure assessment: the direct approach and the indirect approach. With the direct approach, individuals carry personal monitoring devices that measure and record the air pollutant concentrations to which they are exposed for the time period of interest. Personal monitoring devices are available for NO_2, CO, PM, and possibly for ozone and nitric acid. These directly measure the parameters of interest. With the indirect approach, air monitoring data for various microenvironments (outdoors, homes, schools, offices, in cars, etc.) and human time-activity data are collected and used to estimate personal exposure. Personal exposure is estimated as the time-weighted average of the exposure in each microenvironment occupied by the individual for the time period of interest (i.e., the microenvironmental approach). The two approaches may produce slightly different results because humans are sources of certain pollutants, such as PM and ammonia, which may influence the personal monitors. Spatial variations in outdoor and indoor concentrations (i.e., in different rooms) are also likely to cause differences between personal monitors and fixed microenvironmental monitors.

Neither the direct nor indirect measurement approach is practical for a long-term health study of several thousand participants. Personal monitors have typically only been deployed to obtain measurements for short time periods (less than a week). Most people would find it burdensome to wear the personal monitors for long time periods. In addition, for those methods that require laboratory analysis of filters for daily or weekly samples, the analytical costs are prohibitive. Likewise, measuring air concentrations in each location where the study participants spend significant amounts of time (e.g., in their homes) would require a large number of measurement devices. The collection of time-activity diaries on a daily or weekly basis would also be quite burdensome for the participants. Exposure characterization approaches that require fewer measurement devices and infringe less on the study participants are, therefore, desirable.

Fixed-site ambient air monitors are the most cost-effective means of obtaining outdoor pollutant data for community monitoring. Properly sited community monitors can usually provide representative exposure data for the time children spend outdoors. The outdoor concentrations are also an essential input for estimation of indoor pollutant concentrations. Infiltration of outdoor air is the primary determinant of indoor ozone concentration (photocopying machines are the only significant sources of indoor ozone), and, for buildings without indoor combustion devices, it is also the principal source of PM, NO_2, and nitric acid in Southern California. Building air exchange rates, and the type and extent of air filtration, air conditioning, and indoor surfaces are known to affect the indoor/outdoor ratio of pollutants.

A third potential approach for characterizing exposure is the hybrid approach that involves limited measurements and exposure modeling. We believe exposure modeling is a cost-effective technique for extending a limited data base of microenvironmental measurements and time-activity data.

This approach employs questionnaires to provide data on an individual's time-activity patterns and housing/ventilation system characteristics. It requires the development and refinement of indoor/outdoor pollutant relationships that would necessitate collection of microenvironmental measurement data for schools and a sample of homes. In addition, an essential task for establishing the validity of the hybrid approach is evaluation of the exposure model against personal monitoring data. Our overall scheme is depicted in Figure 2.4.1.

Community air monitoring data were used for ozone and PM [a combination of total suspended particulates (TSP) & PM_{10}] in the United States between 1980 and 1992 to determine pre-study exposures of the children who lived in the United States. For the children born in the United States, pre-study exposures were estimated from the mean of the annual average ambient ozone and PM concentrations at the air monitors closest to their residences (accounting for changes in residences).

2.5 STUDY DESIGN AND STATISTICAL APPROACHES

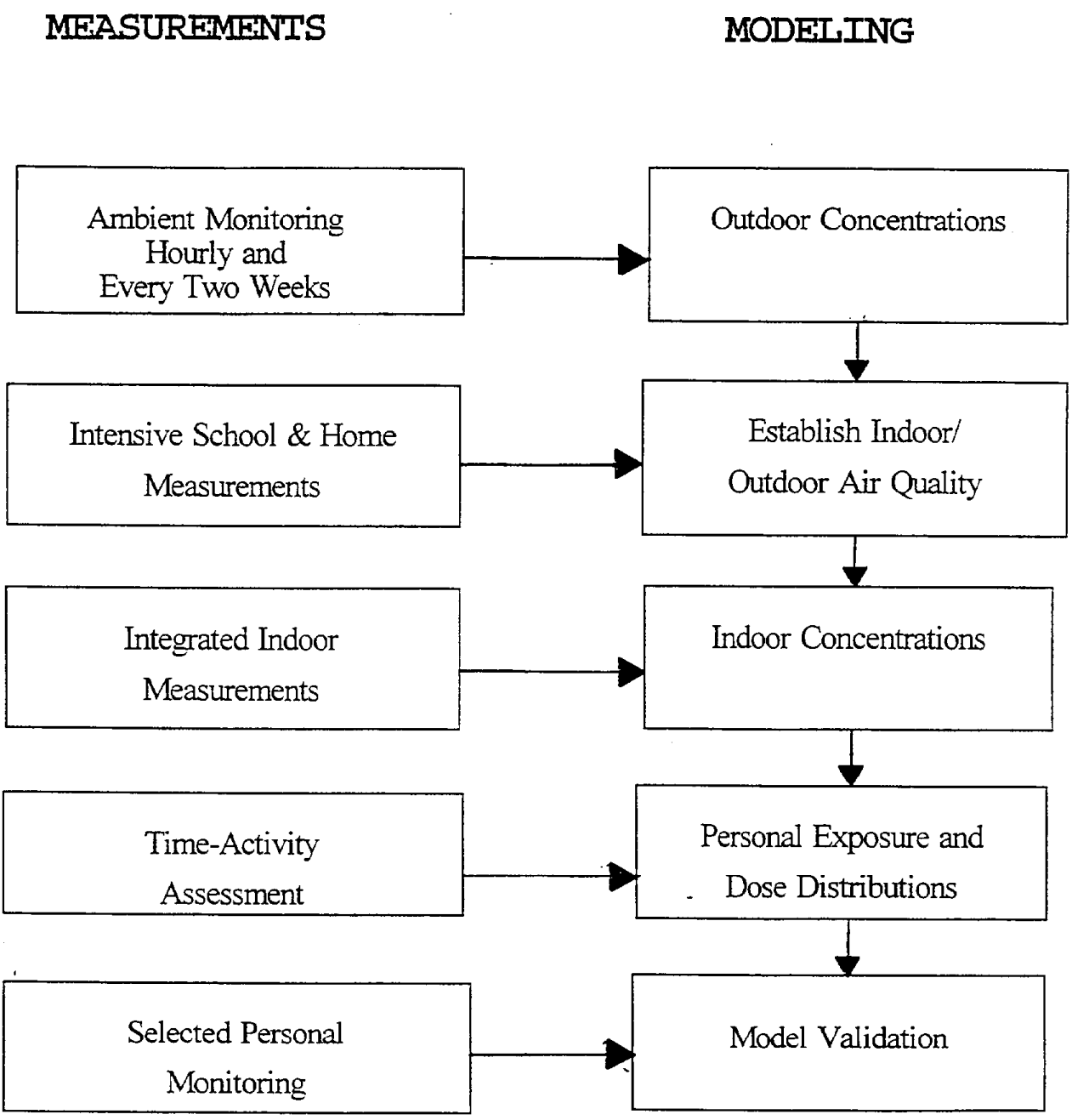
2.5.1 Introduction

Basic issues in study design considered during Phase I included determining the number of pollutants to focus on, the pollutants to consider, the number of communities to involve, the number of subjects in each community to investigate and their times of entry into the study.

The number of pollutants to which the study gave primary focus was determined by the pollution profiles that existed in the communities in the regions of Southern California under consideration along with the potential for health effects based on the scientific literature reviewed for Phase I. We focused on four - 24 hour peak ozone, PM_{10}, vapor-phase strong acids (HNO_3, HCl), and NO_2, but other pollutants, and other measures of exposure, such as 24-hr mean ozone, were considered.

FIGURE 2.4.1

Approach for Determining Exposure



2.5.2 Statistical Methods

SAS statistical packages where used for all analyses except absence monitoring. Absence monitoring was analyzed with programs written by William Navidi. The pulmonary function measurements used in the analyses presented in Section 4 were made in the spring of 1993 and 1994. We focus on forced expiratory volume in one second (FEV_1), forced vital capacity (FVC), peak expiratory flow rate (PEFR), maximum-mid-expiratory flow rate (MMEF) and expiratory flow rate at 25% of vital capacity (FEF_{25}). We consider FEV_1 to be the best overall single measure of lung function because it combines an assessment of both volume and expiratory flow rate. To measure pulmonary function, each subject performed several spirometry maneuvers, stopping after completing three maneuvers deemed acceptable by the software, or after a total of seven maneuvers overall. Since primary interest centers on the subject's ideal performance, we take as the outcome variables the maximum of each of the pulmonary function variables over the maneuvers, after eliminating those maneuvers which were reported by technicians as having been performed improperly.

To adjust for variations in temperature and for spirometer drift, spirometers were calibrated frequently with three-liter volumetric syringes (Jones Instruments, Oakbrook, IL) with digital computerized output during testing sessions. Each calibration consisted of a sequence of syringe maneuvers, for which both the syringe reading and the spirometer reading were recorded. Adjustment of each subject's pulmonary function measurements was performed by multiplying the raw spirometer measurement by the median of the syringe/spirometer FVC ratios among the calibration maneuvers occurring closest in time both before and after the pulmonary function measurement (Linn et al., 1996).

We studied the relationship between lifetime exposure to air pollution and the following morbidity endpoints: active asthma, wheeze, cough, bronchitis, pneumonia, and other chest illness, as reported on the baseline questionnaire. The relationship between morbidity and pollution exposure was studied with unconditional logistic regression. We used SAS logistic procedures with stepwise model selection. For models where exposure was estimated at the individual level, potential confounders were entered one at a time into a model containing age, race, sex, and indicators for the 12 communities. For models where exposure was measured at the community level, indicators for the communities were not included. Those potential confounders that were statistically significant univariately at the 15% level were then subjected to a stepwise procedure to choose the final set of variables for the model. In this process, variables are entered into and removed from the model in such a way that each forward selection step is followed by one or more backward elimination steps. The stepwise selection process terminates if no further variable can be added to the model, or if the variable just entered into the models is the only variable removed in the subsequent backward elimination.

The relationship between pulmonary function and pollution exposure was studied with ordinary least squares regression. Potential confounders were screened in the manner described above, except that the initial model contained variables for height and weight measured on the day of the pulmonary function test, as well as age, race, and sex.

For all pulmonary function and questionnaire-based morbidity endpoints, we considered the following potential confounders: age, race, sex, Hispanic origin, hayfever, current taking of vitamins, membership in a health insurance plan, pets in the household, recent infestation of household by pests, five or more plants in the household, gas stove, gas heat, air conditioning (any type), water damage to home, mold or mildew ever in the home, carpet in subject's bedroom, exposure to passive smoke at home, and family income.

For the pulmonary function endpoints, we also considered asthma (ever diagnosed), active asthma (determined from the questionnaire as described in Section 4.5), and exercise in the 30 minutes prior to the pulmonary function test as potential confounders. For morbidity endpoints we also considered height and weight as potential confounders.

We studied the possible modification of the effect of ambient exposure to air pollutants by physical activity (based on activity questionnaire). Six measures of physical/spatial activity were considered: number of hours outdoors during the last ten school day afternoons, number of afternoon hours outdoors on the last two weekends, number of afternoon hours outdoors during two typical summer weeks, number of hours very physically active outdoors during the last ten school day afternoons, number of afternoon hours very physically active outdoors on the last two weekends, and number of afternoon hours very physically active outdoors during two typical summer weeks. These variables were screened for entry into the individual level model constructed by the method described above. For physical activity variables included in the model, interaction terms with exposure variables were tested for statistical significance.

We performed analyses using both the individual and the community exposure indices. For the community exposure analyses, the historical ambient levels for the years 1986-90 and for the year 1994 were used to measure exposure, while confounders were measured at the individual level. These community mean exposures were included in the regression equations for individual outcomes, together with the individual-level confounding variables. Residual community effects cannot be included in this form of analysis because they are aliased with the community mean exposures, and reflect between-community average effects, i.e. ecologic correlations. For the individual exposure analyses, the lifetime exposure indices were included together with the 12 community indicator variables and individual-level confounders. Exposure associations from these analyses reflect between-individual, within-community effects. A further discussion of single and multiple pollutant models is presented in Section 4. Unless

otherwise indicated, all reported results are for single pollutant models, i.e. only one pollutant was included in the regression model at a time. Multipollutant models were also investigated but the correlations between pollutants made these models uninformative. Use of multipollutant models will be further evaluated for the Phase III analyses.

In reporting our results, we use the term "positive association" to refer to an association between increased exposure and increased pulmonary function or increased incidence or prevalence of morbidity, and the term "negative association" to refer to an association between increased exposure and decreased pulmonary function or decreased incidence or prevalence of morbidity. Thus a positive association is adverse in the case of morbidity, while a negative association is adverse in the case of pulmonary function.

2.5.3 Rationale for the Basic Comparisons

2.5.3.1 Between-Areas Versus Between-Individual Analyses

The Phase II study collected data at both the individual and aggregate levels. On the individual level, we included measurements of outcomes (symptoms, lung function measurements, etc.), residential history, behavioral variables related to exposure, and confounding factors. Exposure information was obtained or estimated, using a combination of ambient data (historical and concurrent measurements), and residential history obtained from questionnaires.

The question arises whether to examine the associations at the individual or the community level or to use some combination of the two. The answer in principle involves making certain trade-offs between issues related to (1) within-community variability in exposure and our ability to assess individual exposure histories, and (2) between-community confounding. To the extent that we might be concerned about lack of comparability of the communities in terms of potential risk factors (particularly unknown or unmeasured risk factors), comparisons should be made within communities at the individual level. On the other hand, if the range of variation of exposures within communities were limited, then within-community analyses would have little power and we would be forced into making a between-community analysis at the aggregate level. Fortunately, there turned out to be a wide variation in lifetime exposures within communities (see figures 4.1.3.1 - 4.1.3.12). Thus we were able to obtain sound results for an individual level analysis. We carried out aggregate analyses as well, and the results may have been affected by between-communities confounding.

2.5.3.2 Cross-Sectional versus Cohort Comparisons

The Phase II study involved the enrollment of a cohort of school children at various ages for immediate cross-sectional examination and for possible longitudinal

follow-up examinations . This approach enabled us to examine age effects in the initial cross-sectional comparison between individuals in different age groups.

Of the cross-sectional and cohort comparisons, we view the cohort comparisons as the more important, mainly because the (possibly) subtle effects of air pollution can be tested by examining changes in health status in the same individual. The demonstration of higher incidence of new disease or higher rates of progression of existing disease in the more heavily exposed subjects provides a more convincing basis for causal inference than does cross-sectional comparisons of prevalence or means (National Academy, 1985).

The cross-sectional comparison between several age cohorts allows us to draw preliminary conclusions about the variation in air pollution effects across age early on in the study but does not allow us to deal with uncontrolled confounding. This makes the conclusions that can be drawn from the cross-sectional Phase II study tentative and possibly different from the results of a prospective follow-up in which these issues can be evaluated more fully. A comparison of age effects in a longitudinal analysis would allow us to separate age and birth cohort effects and thus detect any different patterns of health effects as patterns or levels of air pollution may change.

2.5.4 Community Selection

Community selection was based on exposure patterns and levels, and demographic data of a group of census tracts in 86 communities. The demographic data consisted of about thirty variables for each census tract within 5 kilometers of each monitoring site, including racial and age composition, housing characteristics, and educational levels. These data were derived from the 1990 U.S. Census and available in tabular form.

The basic principle governing the selection of communities was that greater dispersion in the design variables tends to produce more accurate estimates of effects. This principle dictates that the site selection algorithm should be formulated to select a group of communities having widely divergent exposure characteristics.

A second principle we followed was that the communities being compared should be comparable with respect to potential confounding variables. Because the real confounders cannot be predicted in advance and because routine data on them may not be available at the level of census tracts, we were restricted to using variables obtained from the census. Perfect balance between communities on these variables was unlikely to be achievable and if it were, it would not have guaranteed comparability of the real confounders. Hence, we relied on two corollaries of this principle: first, that heterogeneous communities are preferred because they would be more likely to exhibit overlapping distributions of risk factors, thus improving the prospects for making adjustments for confounding in the analysis; and second, replication of exposure profiles

was required, so as to improve the chance of including demographically comparable communities and to allow estimation of residual variance within pollution profiles.

Because of the large number of variables, it was necessary to devise an orderly procedure for considering them. We decided to make an initial selection on the basis of the occurrence of the main pollutants and then use the demographic data to adjust, if necessary.

Two basic approaches were considered for the site selection algorithm. We refer to them as the distance approach and the factorial approach. Both were designed to yield a selection of communities varying greatly in exposure characteristics. Both relied on a common set of measured and interpolated ambient concentrations for 86 air monitoring sites in the study area for the period from 1986 to 1990. Because of differences in the number of locations at which pollutants are measured and the frequency and type of measurements made, we judged that the data were more reliable for ozone than PM, and more reliable for PM_{10} than NO_2, and more reliable for NO_2 than acid vapor. First, as a central point for comparison, the regional average concentration for each pollutant was calculated. Second, the deviation from the regional average concentration of each pollutant was determined for each site. The deviation was expressed as the difference from the average, measured in standard units. Other methods of calculating deviations were considered. Some of them involved transforming the original data to a log scale, others involved expressing deviations as a percentage above or below the average. We found these transformations to have virtually no impact on the final selection of communities.

In the distance approach, the standardized deviations for each site were squared and added to produce a single measure of the "distance" from that site to the central point in the three-dimensional pollution space. Communities whose distances from the center were great were those whose pollution levels were generally well above or well below average. In the factorial approach, each site was categorized as high or low for each pollutant (as determined by whether the level of the pollutant is above or below average at that site identified as "hi/low" or "+/-" in subsequent discussions). For each pattern of low and high exposure, an attempt was made to select two communities whose pollution profiles matched that pattern. The data were computerized and analyzed according to both approaches.

Groups of communities selected by the distance method turned out to be less satisfactory than groups of similar size selected by the factorial method. One disadvantage of the distance method resulted in an overrepresentation of sites exhibiting high ozone levels, reducing the power to determine the specific contribution of ozone level to observed differences in health outcomes between communities.

Replication of study sites was considered highly desirable for both statistical and other study design reasons mentioned previously. We attempted to select the two sites

representing similar pollution profiles with as much geographical distance between them as possible but discovered that this was often not possible because the same pollution profile tended to occur in the same local areas. Our pairs, therefore, tend to be geographically adjacent.

The final set of communities selected for study are presented in Table 2.5.4 along with their deviations from regional average long-term pollutant concentrations.

2.5.5 Rationale for Replication of Exposure Profiles

After studying the pollution patterns and the available communities, we identified seven distinct air pollution profiles based on the four pollutants of interest. We attempted to choose more than one community for each pollution profile when that was possible.

Two communities with similar exposure profiles could differ systematically in health outcomes, for reasons such as differences in confounding factors (e.g. demographics), differences in the degree of systematic measurement error such as might be caused by machine malfunction, and differences in biases due to events such as viral epidemics, local sources of pollution or local school administrative practices. Replication of communities was necessary to separate the effects of pollution from the effects of these other factors.

Furthermore, without replication it would have been impossible to estimate accurately at the aggregate level the proportion of variation in health effects due to the pollutants under study as opposed to other unmeasured factors. We recognize that our replication efforts are imperfect, due to the fact that not all profiles could be replicated. The usefulness of replication will ultimately depend on the relative sizes of the between-profile and within-profile variations in confounding factors. By looking at the variation of health measures within two communities sharing the same air pollution profile, we can estimate the contribution of the unmeasured confounding of the health variables to the overall variability between communities, i.e., we can determine the contribution of the air pollution in accounting for the health effects when analyzing variation within the same profile.

Table 2.5.4 Deviation* from Regional Average Long-Term Concentrations in Selected Communities

Communities Selected	Peak O_3	PM_{10}	Acid Vapor	NO_2
San Dimas	+1.8	+1.2	+1.3	+1.1
Upland	+1.5	+2.2	+1.5	+1.5
Mira Loma	+1.6	+2.8	-0.9	+0.7
Riverside	+1.6	+2.8	-0.9	+0.7
Alpine	+0.9	-0.5	-0.8	-0.4
Lancaster	+0.5	+0.2	+1.5	-0.7
Long Beach	-0.8	+0.4	+0.7	+1.5
Lake Elsinore	+1.0	+1.1	+0.4	-0.5
Lake Arrowhead	+1.8	-0.4	+0.5	+0.1
Santa Maria	-1.5	-1.1	-1.1	-1.0
Atascadero	-0.1	-1.1	-1.1	-1.0
Lompoc	-1.3	-1.0	-1.1	-1.4

* Deviation = $\frac{\text{[site mean - regional mean]}}{\text{regional standard deviation}}$.

2.6 REFERENCES

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3. CHARACTERIZATION OF AIR POLLUTION EXPOSURE

3.1 THE EXPOSURE ASSESSMENT METHODOLOGY

As noted in Section 2.4, the approach for characterization of exposure to air pollution incorporated an ambient air monitoring program, a school ozone monitoring program, a residential monitoring program, and a personal ozone monitoring program. This section provides an overview of the exposure assessment methodology. A detailed description of the methodologies is provided in Lurmann et al., 1994.

3.1.1 The Ambient Air Monitoring Network

The objective of the ambient air monitoring program was to obtain continuous measurements of ozone, NO_2, and PM_{10} concentrations on an hourly basis in all of the communities. An additional objective was to obtain integrated measurements of $PM_{2.5}$ mass, $PM_{2.5}$ sulfate, $PM_{2.5}$ nitrate, $PM_{2.5}$ chloride, and $PM_{2.5}$ ammonium aerosol, and gaseous nitric acid, hydrochloric acid, formic acid, and acetic acid for determination of seasonal and annual average concentrations.

Ambient air quality monitoring stations were established in the 12 communities shown in Figure 3-1. Seven of the air monitoring stations were existing sites where air pollution control agencies monitored hourly ozone, hourly NO_2 (actually NO_x-NO), and/or daily PM_{10}. The existing sites included the San Luis Obispo County Air Pollution Control District's (SLOAPCD) site in Atascadero, the ARB's site in Santa Maria, the South Coast Air Quality Management District's (SCAQMD) sites in North Long Beach, Lancaster, Upland, and Lake Elsinore, and the San Diego County Air Pollution Control District's (APCD) site in Alpine. Five new air monitoring sites were established for the study. These new sites were located at the UC Riverside Agricultural Station in Riverside, Jurupa Valley High School in Mira Loma, Rim of the World High School in Lake Arrowhead, Cabrillo High School in Lompoc, and Gladstone Elementary School in San Dimas.

All existing sites monitored hourly ozone with EPA-approved instrumentation. Two existing sites did not monitor NO_2 and only one existing site, Long Beach, monitored PM_{10} on a continuous basis. Existing and new stations were equipped to measure hourly ozone using UV instruments, hourly NO_2 (as NO_x minus NO) using chemiluminescent instruments, and hourly PM_{10} mass using the taped element oscillating microbalance (TEOM) instrument (Patashnick and Rupprecht, 1991). A new aerosol/acid sampler was developed to collect 2-week integrated samples of $PM_{2.5}$ mass, $PM_{2.5}$ sulfate, $PM_{2.5}$ nitrate, $PM_{2.5}$ chloride, and $PM_{2.5}$ ammonium aerosol, and gaseous nitric acid, hydrochloric acid, formic acid, and acetic acid. This development effort was needed because the existing aerosol/acid technologies were designed for 4- to 24-hr sampling intervals and the use of short-interval sampling was prohibitively expensive in

a long-term study. Thus, a major effort was undertaken to develop an acceptable two-week aerosol/acid sampler. The air monitoring was initiated at the times shown in Table 3-1.

3.1.2 The School Ozone Monitoring Program

Virtually no ozone or PM concentrations data were available for Southern California schools at the start of this program. Since children spend a significant amount of time at schools, characterization of air quality at schools was considered essential. The Phase I assessment found less was known concerning indoor/outdoor relationships for ozone than PM or NO_2, so ozone monitoring in schools was given priority for Phase II. PM and possibly NO_2 monitoring in schools was scheduled for Phase III of the study.

The objectives of the school ozone monitoring program were (1) to collect sufficient indoor and outdoor ozone concentration data at schools to classify all schools with respect to their indoor/outdoor ozone relationships, (2) to characterize the variability of the ozone indoor and outdoor concentrations between schools, and (3) to collect outdoor ozone concentration data at schools to characterize the within-community variation in ozone air quality. Since there were 50 schools at the beginning of Phase II, and since numerous schools have a mixture of building types and heat/ventilating system types, it was clear that only a subset of the classrooms visited by students in the study could be monitored. In fact, in planning the study, it was determined that it would be prohibitively expensive to use continuous hourly monitors in all schools. An alternate sampling approach was developed that used a combination of continuous hourly instruments in a few classrooms and inexpensive portable timed exposure diffusion (TED) samplers in two classrooms in each school.

A quiet, portable, battery-operated TED sampler was developed specifically for classroom sampling for this study. The sampler provides controlled airflow across two single-ended Ogawa badges with sodium nitrite-coated filters for measuring ozone by the Koutrakis method (Koutrakis et al., 1993). The sampling program consisted of measuring ozone concentrations inside two classrooms and at one outdoor location, usually on the roof, of each school for 2 weeks in Fall 1993 and 2 weeks of Spring 1994. During each week, each TED sampler collected a 4-day integrated ozone sample for the hours children attended schools (usually 8:00 a.m. to 3:00 p.m.). These data were used to address both objectives of the school sampling program.

In addition, limited continuous monitoring of hourly ozone was conducted in selected classrooms. The objectives of the continuous monitoring were to assess the validity of the integrated ozone obtained using the TED sampler, and to investigate the extent of short-term variations in indoor/outdoor ratios, and the extent of week-to-week

and seasonal variations in the indoor/outdoor ratios. Continuous ozone monitors were installed and operated in four different types of classrooms for periods ranging from 4 weeks to 5 months to address these measurement objectives.

3.1.3 The Residential Air Monitoring Program

Virtually no indoor ozone data and only small amounts of PM data were available for Southern California residences. A moderate amount of indoor and outdoor NO_2 data were available for Southern California residences. The primary objective of the residential monitoring program was to provide indoor residential, outdoor residential, and personal exposure assessments for 150 children in the study. The information collected in the program was intended to enhance the overall characterization of exposure of all children in the study. The enhancement will be accomplished by using the data collected in the residential study to develop and refine exposure models. The specific objectives of the study were to:

- Assess the applicability and limitations of the REHEX model and modify it, as needed, to address large-scale cross-sectional and prospective epidemiological investigations.

- Validate the model using exposure data for a representative cross section of the children in the study.

- Compare the exposure estimates derived from interpolation of fixed-site ambient monitoring data to exposure estimates derived from microenvironmental and personal sampling data.

- Expand the current database on human exposure to current levels of indoor air contaminants that have been hypothesized to produce pulmonary health effects.

- Investigate relationships between indoor/outdoor ratios, housing characteristics, and air exchange rates for a representative sample of study participants' homes.

- Construct a reliable and cost-effective long-term sampler for indoor/outdoor measurements of fine particles and airborne acids.

Owing to resource constraints, measurements of $PM_{2.5}$, PM_{10}, and formaldehyde were only planned for a subset of the homes. Personal ozone sampling was to be conducted on the children on the days the homes were sampled. However, prior to initiating the personal sampling portion of the program, the study team needed to identify and evaluate a suitable personal ozone sampler. Refinement and testing of the exposure model will follow the residential data collection effort.

3.1.4 The Personal Ozone Monitoring Program

The objectives of the personal ozone monitoring program were to evaluate personal monitoring devices and collect personal ozone samples and concurrent time-activity diaries for time periods of 7 to 24 hours. The sampler evaluation data were needed to establish the credibility of the devices. The personal ozone data were needed to evaluate and refine the indirect exposure assessment procedure which is embodied in the exposure model.

At the beginning of Phase II, there were no acceptable personal ozone monitoring devices; however, the Harvard group was working on the development of several new personal sampling devices. Since the ARB study did not include personal sampler development work, the schedule for the program was largely determined by Harvard. Fortunately, the Harvard group developed several prototype samplers in time for laboratory testing in Spring 1994 and field testing in Summer 1994. The laboratory and field evaluations indicated an active personal ozone sampler was sufficiently accurate and precise for use in the study. In October 1994, 580 personal ozone samples were collected on sixth grade students in two communities who attended four different schools.

The approach for the methods evaluation involved close coordination with the Harvard group developing the personal ozone samplers, interference testing of the Koutrakis badge included in the TED sampler evaluation, separate chamber testing of the passive badge and an active hollow tube personal sampler, and field testing of both the active and passive badges on 60 children in an "all outdoor" experiment in Riverside. The "all outdoor" experiment (Experiment No. 1) was completed in July 1994.

The intended approach for monitoring of personal ozone levels consisted of three more experiments. In the second experiment, 140 students were monitored on one weekday for personal ozone levels during school (8:00 a.m. to 2:00 p.m.) and after school (2:00 p.m. to 6:00 p.m.). Ozone levels were measured indoors and outdoors at the schools and trained observers recorded their time-activity while at school. Diaries were given to the children for the afternoon hours. Experiment No. 2 provided the opportunity to evaluate the exposure model with almost ideal inputs during the hours the children were in school and the opportunity to test the diary instrument prior to using it on a large group of students. It also provided personal exposure data to assess the relative important of at-school exposure compared to after-school exposure. Data from this experiment were used to evaluate the adequacy using I/O ratios and time-activity data to predict personal exposure, which is central to the overall exposure characterization approach.

The research plan called for a third experiment involving collection of personal monitoring data and diary information on children while their homes were being

monitored in the residential study. This experiment was not carried out because the residential program sampled homes from February to November 1994 and the personal monitoring device was not available for use in the program until September 1994. This important experiment was postponed until Phase III.

A fourth experiment, which involved monitoring of 140 students for 9 hours on one weekday and one weekend day, was completed. The students used a self-reported hourly diary. These data were used to evaluate exposure model performance on weekdays and on weekend days. The experiment differed from the original Phase II plan in that each child was monitored for 2 days rather than 6 days. The experiment was curtailed because the ozone season ended. The smaller sample size and unusually low ambient ozone levels during the sampling limits the utility of the data for model evaluation and for extrapolation to all participants in the study. Nevertheless, the program demonstrated the feasibility of conducting personal ozone sampling with the active personal ozone samplers and provided initial data for testing the ozone exposure model.

3.2 AMBIENT AIR QUALITY IN THE 12 COMMUNITIES

This section describes the ambient air quality data used for the study design, the current air quality conditions in the 12 communities, and the current pollutant profiles for the 12 communities. Data for the first full year of monitoring in the communities are used to characterize current air quality conditions. Because meteorological conditions vary from year to year, and because air quality control programs are in place to reduce emissions over time, the current air quality pattern should be considered a snapshot of a long-term pattern. The pattern is expected to vary over time and one year is the minimum time interval at which to examine the pattern in a long-term study.

3.2.1 Historical Air Quality Data Used for the Study Design

As discussed in Section 2.5, measured and interpolated ambient pollutant concentrations for 86 monitoring sites during 1986 to 1990 were used in the community selection process. The process included a factorial approach to community selection based on estimated deviation of the long-term concentrations from the regional mean concentrations. It is presently unresolved whether the health endpoints associated with ozone exposure are related to exposure to high peak concentrations or the cumulative ozone exposure. For the study design, the long-term average of the daily 1-hr maximum ozone, 24-hr NO_2, and 24-hr PM_{10} concentrations were used. It should be noted that, among the sites selected, the 1-hr peak ozone concentrations correlated well with all daytime measures of ozone exposure (e.g., 10:00 a.m. to 6:00 p.m.).

The 1986 to 1990 historical ambient air quality database provided reasonably good spatial coverage for ozone, PM_{10}, and NO_2, but poor spatial coverage for acidic species. Ozone was measured at most of the 86 sites, and NO_2 and PM_{10} or TSP (total

suspended particulate) were measured at about a third of the sites. Interpolation was used to estimate ambient concentrations at locations where pollutant concentrations were not measured. In addition, an approximate relationship between PM_{10} and TSP mass was used to estimate PM_{10} concentrations (PM_{10} = 0.55*TSP) in locations and periods where only TSP data were available. Ambient nitric acid concentration data were significantly more limited: data were collected at eight sites in the South Coast Air Basin (SoCAB) during 1986 (Solomon et al., 1988; Eldering et al., 1991); at three SoCAB sites, plus several sites outside the SoCAB since 1988 as part of the California Acid Deposition Program; and at Simi Valley during 1989 to 1990 (Thompson et al., 1991; Spengler, 1992). Less is known about formic and acetic acid concentrations in Southern California: Solomon et al., (1988) collected formic and acetic acid data at eight sites from May through December 1986; in addition, some measurements were made during limited periods (i.e., on high-ozone days during intensive field studies such as Southern California Air Quality Study (SCAQS) in 1987) or at only one or two sites. Formic and acetic acid data were not used for the study design.

The USC investigators were not able to gain access to the students in four of the 12 communities selected in Phase I. Nevertheless, alternate communities with similar historical pollutant profiles were identified and selected. The difference are as follows:

- San Dimas is a neighboring community to Glendora (originally selected) and was expected to have similar pollutant characteristics because of its close proximity (4 km) to Glendora.

- Mira Loma is located 10 km west of Rubidoux (originally selected) and was expected to have pollutant characteristics similar to those of Rubidoux due to their close proximity.

- Lake Elsinore is located 10 km south of Perris (originally selected). The 1986-1990 data indicate Lake Elsinore had mean daily maximum ozone concentrations (83 ppb) that were similar to those measured at Perris (81 ppb). Lake Elsinore was expected to have similar NO_2 and PM_{10} concentrations to Perris because of its close proximity.

- Atascadero is located 110 km north of Santa Barbara (originally selected) and has somewhat similar pollutant characteristics. The 1986-1990 historical data indicate ozone concentrations were higher in Atascadero than Santa Barbara, and NO_2 and PM_{10} concentrations were lower in Atascadero than Santa Barbara.

3.2.2 Current Air Quality

This section discusses the status of the air quality data for the 12 communities through May 1995, presents a summary of the data, and discusses similarities and differences in the data in the various communities. The measurement methods used to

collect these data, and the data processing and validation procedures used are discussed in detail in Lurmann et al., 1994.

3.2.2.1 Status of ambient air quality monitoring

The ambient air monitoring was conducted at existing ARB or local air pollution control agency sites in seven communities. Sonoma Technology, Inc. (STI) established five new monitoring sites for the study in the communities of Lompoc, San Dimas, Mira Loma, Riverside, and Lake Arrowhead. As shown in Table 3-1, ozone was monitored by government agencies at the seven existing ARB or local agency sites. NO_2 was monitored by government agencies at six of the seven existing ARB or local agency sites. Continuous PM_{10} (by TEOM) was monitored by the SCAQMD at the existing Long Beach site. STI purchased, installed, and operated the additional continuous monitoring equipment (5 ozone monitors, 7 NO_2 monitors, and 11 PM_{10} monitors) needed for the 12-station network. As shown in Table 3-1, continuous monitoring for ozone, NO_2, and PM_{10} began during Fall 1993 in most communities; the exceptions were at Lake Arrowhead and Alpine where lack of access to the sites delayed the initiation of monitoring until February and March, 1994. STI installed and operated Two-Week Samplers (TWSs) at all of the sites. Measurements of acidic species and $PM_{2.5}$ aerosols with the TWS began in December 1993 in all of the communities except Lake Arrowhead and Alpine, where they began in February and March, 1994.

As part of the planned transition from Phase II to Phase III of the study, the ambient air monitoring responsibilities were transferred from STI to the ARB (working in collaboration with local air quality districts) during 1994. Operation of the new continuous monitoring equipment installed at existing government agency sites was transferred to ARB or local agencies early in 1994 (except at the Santa Maria and Alpine sites). Overall operation of the network was transferred to the ARB late in 1994. Specifically, the continuous monitoring operations at the five new stations were transferred to the ARB in October 1994 and the TWS field operations at all of the stations were transferred to the ARB in December 1994. All preparation and analysis of TWS filter substrates was conducted by the LAREI laboratory throughout 1993 and 1994.

Data for the 12-month period from January 1 to December 31, 1994 are reported here. Operations at Lake Arrowhead and Alpine started late, and complete data are not available for several other sites, as described below. In general, data recovery was excellent in both the continuous and TWS networks. Of the operating months, 95 percent of the ozone, 92 percent of the NO_2, and 95 percent of the PM_{10} monthly averages were recovered. For the TWS, 97 percent of the $PM_{2.5}$ mass, acidic species, and inorganic ion data were recovered. With the TWS, nine species concentrations are measured: $PM_{2.5}$ mass; gaseous nitric, hydrochloric, acetic, and formic acids; and $PM_{2.5}$

nitrate, sulfate, ammonium, and chloride inorganic ions. On an individual species basis, 98 percent of the 1413 individual species concentration measurements possible were recovered.

The completeness and validation level of the first-year ambient concentration data set used to prepare this report is as follows:

- Quality-controlled ozone, NO_2, and PM_{10} data for Atascadero (operated by the San Luis Obispo APCD) were archived through December 1994.

- Quality-controlled ozone data for Santa Maria (operated by the ARB) were archived through October 1994; NO_2 and PM_{10} data (collected by STI) were archived from February through December 1994 at Santa Maria.

- Quality-controlled ozone, NO_2, and PM_{10} data for the existing sites operated by the South Coast AQMD (Lancaster, Long Beach, Upland, and Lake Elsinore) were archived through December 1994.

- Quality-controlled ozone and NO_2 data for Alpine (operated by the San Diego APCD) were archived through September 1994; and quality-controlled PM_{10} for Alpine were archived through October 1994.

- Quality-controlled ozone, NO_2, and PM_{10} data for Lompoc, San Dimas, Mira Loma, Riverside, and Lake Arrowhead (the five new sites operated by STI through October 1994 and operated by local districts or ARB thereafter) were archived through December 1994. Continuous data are not available for Lake Arrowhead until March 1994.

- SCAQMD and STI experienced some problems with both the NO and NO_x channels of the nitrogen oxides monitor at Lake Elsinore; the SCAQMD subsequently changed the monitor at that site. These problems resulted in significant data gaps in NO_2 concentrations at Lake Elsinore during April and May. Also, PM_{10} data were not available at Lake Elsinore until February 1994.

- Quality-controlled $PM_{2.5}$ mass, $PM_{2.5}$ ionic species concentrations, and acidic species concentrations from the TWSs were archived through December 1994. As noted above, TWS data for the first few months of 1994 were not available for Lake Arrowhead and Alpine due to delays in startup.

- Problems with the laboratory balance resulted in invalid $PM_{2.5}$ mass concentration data for samples collected from mid-February to mid-March in all of the communities. Valid $PM_{2.5}$ ionic species concentrations and acidic species concentrations were obtained for these two sampling periods.

- Four 2-week acid/aerosol samples were invalidated during the year due to TWS pump failures. The pump problems occurred in Lake Elsinore, Lancaster, and Mira Loma. The pump manufacturer provided improved pump seals, which were installed in all TWS pumps to address the problem. Overall, the TWS performed well; only four out of 157 samples were lost due to sampler problems in the first full year of operation.

- The organic acid samples were invalid for one period at Upland, and chloride data were invalid for two periods at Lancaster and one period at Lake Arrowhead due to minor laboratory problems.

3.2.2.2 Summary of the air quality data for the 12 communities

Ozone

The air quality measurements collected in the 12 communities are compared below. For epidemiologic purposes, comparison of the relative differences is most important. For this analysis of differences between communities, a high average concentration is defined as one that is 0.5 standard deviations above the mean for the communities. Similarly, a low concentration is defined as one that is less than 0.5 standard deviations from the community mean concentration. An average concentration is within ±0.5 standard deviations from the community mean concentration.

The 1994 annual and May to September mean values for the following 12 ozone exposure metrics are shown in Tables 3-2 through 3-5, and in Figures 3-2 through 3-7:

- 24-hr average concentration
- 10:00 a.m. to 6:00 p.m. average concentration
- 2:00 p.m. to 6:00 p.m. average concentration
- 1-hr daily maximum concentration
- Hours per month with ozone concentrations ≥ 60, 90, 120, and 150 ppb
- Concentration-hours per month (ppb-hrs) with ozone concentrations ≥ 60, 90, 120, and 150 ppb

These metrics were selected to characterize both the average and peak concentrations. Note the relative differences between sites varies for the different exposure metrics. Also, the seasonal variation in ozone levels is high, as shown in Figure 3-8, with the highest concentrations generally occurring from May to September. The ozone data from the May to September period are important not only because they characterize the high ozone season, but also because the data sets are complete in all of the communities and, therefore, are valid for between community comparisons. Children are also likely to spend more time outdoors during this extended summer period and receive their highest dose during this period.

The annual 24-hr average data indicate Lake Arrowhead and Long Beach have the highest (71 ppb) and lowest (19 ppb) ozone concentrations, respectively, of the 12 communities. The next highest annual 24-hr average ozone was observed at Alpine (44 ppb), which like Lake Arrowhead, is a high elevation site. The May-September data indicate the mean 24-hr ozone concentration at Lake Arrowhead (82 ppb) was almost 70 percent higher than the next highest community (Lancaster, 48 ppb) in the extended summer. Most of the other communities have May to September average ozone levels between 30 and 47 ppb.

The data for the daytime ozone metrics, such as the 1-hr maximum, 2:00 p.m. to 6:00 p.m. average, and 10:00 a.m. to 6:00 p.m. average, show a somewhat different pattern than those for the 24-hr average. Lake Arrowhead stands out as the highest ozone station (137 ppb 1-hr max in May to September), and San Dimas, Upland, Mira Loma, and Riverside have high daytime ozone levels (108 to 126 ppb 1-hr max in May to September). On an annual basis, the daytime levels at Alpine are also quite high (77 ppb average 1-hr max). Long Beach, Lompoc, Santa Maria, and Atascadero have low average daytime ozone concentrations (e.g., 35 to 58 ppb 1-hr max in summer). The ratio between the highest and lowest average daytime concentrations in the communities is about three.

Examination of the average number of hours per month with ozone concentrations above 60, 90, 120, and 150 ppb shows large differences between the less polluted and more polluted communities. During the May to September period, Long Beach, Lompoc, Santa Maria, and Atascadero have less than 0.2 hours per month above 120 ppb, whereas Upland, San Dimas, Mira Loma, Riverside, and Lake Arrowhead have from 29 to 109 hours per month above 120 ppb. Lake Arrowhead clearly stands out with 475, 233, 109, and 40 hours per month with ozone above 60, 90 ,120, and 150 ppb, respectively. The Lake Arrowhead community has almost twice as many hours per month above these thresholds as the other high ozone communities. The concentration-weighted (i.e., sum of the hourly concentrations) hours per month data essentially show the same differences between sites as the hours per month data. During the May to September period, Upland, San Dimas, and Riverside have about 8000 ppb-hrs per month with ozone above 120 ppb and Lake Arrowhead has 16,000 ppb-hrs. Mira Loma and Lake Elsinore are moderate having 4200 and 2600 ppb-hrs per month with ozone above 120 ppb. All other sites have less than 520 ppb-hrs per month during May to September.

There are also day-of-week variations in ambient ozone concentrations. Figure 3-9 and 3-10 show that the mean 10:00 a.m. to 6:00 p.m. ozone concentrations are slightly higher on Saturdays than other days in most of the SoCAB communities (e.g., at Upland, San Dimas, Mira Loma, Riverside, Lake Arrowhead, and Alpine). Children probably spend more time outdoors on weekend days than other days, so their exposures may be enhanced by the day-of-week pattern. However, the low ozone communities outside of the SoCAB do not show any significant day-of-week variations.

Ozone concentrations are known to vary significantly throughout the day. Figure 3-11 shows the diurnal profile of the ozone concentration at the 12 communities. The profiles are consistent with the expected ozone diurnal profile: concentrations rise in the late morning after the heavy traffic subsides and peak in the afternoon. Ozone concentrations decline in the late afternoon and are low throughout the night. The profiles at the four Los Angeles Basin sites are particularly sharp, with peaks at 1:00 to 2:00 p.m., and concentrations below background in the nighttime and early morning as the result of NO titration. The concentrations between about 2:00 to 5:00 p.m., when children are out of school and likely to be outside, are still relatively high. Lake Arrowhead is unusual in that the peak occurs at about 4:00 p.m. in the afternoon. The peak ozone occurs later in the afternoon at Lake Arrowhead than other sites because of the time required to transport ozone to this downwind site. Also, because of the lighter traffic in Lake Arrowhead, there is less NO in the late afternoon to titrate the ozone. The high concentrations in the late afternoon may affect children's exposure as they are more likely to be outdoors during these after-school hours.

Nitrogen Dioxide

The annual and May to September 24-hr average ambient NO_2 concentrations in 1994 in the 12 communities are shown in Table 3-6 and in Figure 3-12. The annual data indicate Atascadero, Lompoc, Santa Maria, Alpine, and Lake Arrowhead have low (< 14 ppb) ambient NO_2 levels. Lancaster and Lake Elsinore have moderate (19 to 20 ppb) NO_2 levels. San Dimas, Upland, Long Beach, Mira Loma, and Riverside have high (33 to 41 ppb) NO_2 levels. Upland has the highest levels, with an annual average concentration of 41 ppb. The May to September data exhibit a spatial pattern similar to the annual data. However, monthly data shown in Figure 3-13 indicate most communities experience the highest NO_2 levels between August and January, and lowest levels in March through May.

Figure 3-14 shows the diurnal profile of the nitrogen dioxide concentration in the 12 communities. At many of the sites there is a depression in the nitrogen dioxide concentration during the mid-afternoon. The highest concentrations occur during the mid-morning and late afternoon and evening. The higher concentrations in the late afternoon are significant because these are the hours that the children are most likely to be outdoors.

Particulate Matter Less than 10 μm in Diameter

Table 3-7 and Figures 3-15 and 3-16 show the 1994 PM_{10} concentrations for the 12 communities measured by the TEOM monitors. The measurements indicate 24-hr average PM_{10} concentrations are lowest in Lompoc (14.5 $\mu g/m^3$) and also relatively low in Atascadero, Santa Maria, Alpine, and Lake Arrowhead (20 to 23 $\mu g/m^3$). Moderate to high (28 to 38 $\mu g/m^3$) annual average (24-hr) PM_{10} concentrations were observed in Lancaster, San Dimas, Upland, Long Beach, Riverside, and Lake Elsinore. The highest

PM_{10} concentrations were observed at Mira Loma, where the 1994 annual average concentration was 55 $\mu g/m^3$. The daytime PM_{10} exposure metrics (1-hr max, 2:00 p.m. to 6:00 p.m., and 10:00 a.m. to 6 p.m. averages) mostly show a similar spatial pattern. For example, the lowest average daily 1-hr maximum concentration was 30 $\mu g/m^3$ at Lompoc and the highest average daily 1-hr maximum concentration was 116 $\mu g/m^3$ at Mira Loma. However, Lancaster had the second highest (77 $\mu g/m^3$) average daily 1-hr maximum concentration and only the sixth highest (30 $\mu g/m^3$) 24-hr average concentration; similarly, Lake Arrowhead had the fourth highest (63 $\mu g/m^3$) average daily 1-hr maximum concentration and only the ninth highest (22 $\mu g/m^3$) 24-hr average concentration. In addition, the 1994 PM_{10} data show a surprisingly large difference in concentrations between Riverside (34 $\mu g/m^3$) and Mira Loma (55 $\mu g/m^3$), which are only 15 km apart. The Riverside PM_{10} levels are also quite low compared to the historical PM_{10} levels at the Rubidoux site. The low PM_{10} concentrations at Riverside are being investigated in more detail in Phase III. A collocated TEOM was installed at this site to investigate whether the current monitor is operating properly.

The additional exposure metrics for PM_{10}, presented in Table 3-8 and Figures 3-17 and 3-18, show an even greater spread between the sites. For example, the average number of hours per month with PM_{10} above 50 $\mu g/m^3$ is 4 in Lompoc and 160 in Mira Loma; the average number of hours per month with PM_{10} above 150 $\mu g/m^3$ is 0.3 in Lompoc and 18 in Mira Loma. The relative ranking of sites based on the number of hours above various concentration thresholds is generally similar to the ranking based on the average daily 1-hr maximum concentrations. It should be noted that sampling started late in Lake Arrowhead (3/11/94) and Alpine (2/25/94). PM_{10} concentrations in January and February may be somewhat higher than the rest of the year (see Figure 3-16); therefore, the estimates for Lake Arrowhead and Alpine may be lower than the true annual average for these sites. This will be investigated further when the 1995 to 1996 data are available.

There are some concerns about the performance of the TEOM monitors because volatile PM_{10} constituents, such as ammonium nitrate and certain organic compounds, are not sampled properly by the TEOM. The volatile constituents appear to be lost from the TEOM filter because the sampler is operated at a higher than ambient temperature (50°C) (see Lurmann et al., 1994, Appendix B, and Allen et al., 1995). The relative amounts of volatile PM_{10} constituents are expected to vary between communities and, therefore, the TEOM data may be spatially biased. The bias (under-reporting of mass) is probably largest in Mira Loma and Riverside, which generally have high ammonium nitrate concentrations. A detailed evaluation of the TEOM measurements, including a comparison of collocated TEOM and HiVol PM_{10} monitors in three communities, is presented in Appendix B. Notwithstanding its problems, the TEOM is still the best continuous PM_{10} monitor currently available and it may be possible to modify operating procedures or develop adjustment procedures to reduce the bias in the PM_{10} data.

Figure 3-19 shows the diurnal profile of the PM_{10} concentration at the 12 communities. PM_{10} has a much less distinct diurnal profile than ozone and nitrogen dioxide at most sites. This pattern simplifies the exposure assessment. However, Mira Loma, Upland, Lancaster, and Atascadero have a bi-modal distribution with peaks in the morning and late afternoon, which probably reflects the influence of traffic patterns.

$PM_{2.5}$ Aerosol and Gaseous Acids

A tabulated summary of the 1994 annual average concentrations of $PM_{2.5}$ mass; $PM_{2.5}$ nitrate, sulfate, ammonium, and chloride ions; and gaseous nitric, hydrochloric, acetic, and formic acids is given in Table 3-9. Figures 3-20 through 3-24 illustrate community variations for these species. Temporal variations in these pollutants are illustrated in Figures 3-25 through 3-36. The 1994 annual (i.e., average of the 2-week integrated measurements) data indicate Lompoc, Santa Maria, Atascadero, Lancaster, and Alpine have low (7 to 10 $\mu g/m^3$) $PM_{2.5}$ concentrations; Long Beach, Lake Arrowhead, and Lake Elsinore have moderate (11 to 16 $\mu g/m^3$) $PM_{2.5}$ concentrations; and San Dimas, Upland, Riverside, and Mira Loma have high (22 to 31 $\mu g/m^3$) $PM_{2.5}$ concentrations. As was the case for PM_{10}, Mira Loma had the highest $PM_{2.5}$ concentration (32 $\mu g/m^3$). These data indicate there is approximately a factor of five difference in $PM_{2.5}$ concentrations between the least and most polluted communities.

Of the measured $PM_{2.5}$ ions, nitrate was the most abundant at all of the sites, followed by ammonium, sulfate, and then chloride. Nitrate levels ranged from 0.8 $\mu g/m^3$ at Lompoc to 13.5 $\mu g/m^3$ at Mira Loma. Ammonium levels ranged from 0.4 $\mu g/m^3$ at Lompoc to 4.9 $\mu g/m^3$ at Mira Loma. Sulfate levels ranged from 0.7 $\mu g/m^3$ at Atascadero to 2.9 $\mu g/m^3$ at Mira Loma. The spatial variations in sulfate were considerably less than nitrate and ammonium. $PM_{2.5}$ chloride levels were extremely low (<0.3 $\mu g/m^3$) at all sites. These four ions accounted for 36 to 70 percent of the $PM_{2.5}$ mass. The ion balance suggests that the nitrate is primarily bound as ammonium nitrate (NH_4NO_3) and that the sulfate is ammonium sulfate [$(NH_4)_2SO_4$)]. Ammonium nitrate accounted for 12 to 43 percent of the $PM_{2.5}$ mass. More than 40 percent of the $PM_{2.5}$ was ammonium nitrate in the Mira Loma and Riverside communities. Ammonium sulfate accounted for 9 to 23 percent of the $PM_{2.5}$. The ion balance [$NH_4/18 \div (NO_3/62 - 2*SO_4/96 - Cl/35)$] was within ± 10 percent at all of the sites measuring significant concentrations, which indicates the ion data are consistent. The deviations in the ion balances were higher at the least polluted sites, but the ion measurements at these sites were often close to or below the detection limits.

The total gaseous acid concentrations (i.e., the sum of nitric acid, hydrochloric acid, formic acid, and acetic acid) were low (2 to 4 ppb) at Atascadero, Lompoc, and Santa Maria; moderate (5 to 8 ppb) at Long Beach, Lake Elsinore, Lake Arrowhead, Lancaster, and Alpine; and high (10 to 13 ppb) at San Dimas, Upland, Mira Loma, and Riverside. Organic acids were more abundant than inorganic acids at all sites. Formic plus acetic acids were 52 to 71 percent of the total gaseous acid concentrations. Nitric

plus hydrochloric acids were 29 to 48 percent of the total gaseous acid concentrations. Acetic acid was the most abundant individual acid, with concentrations ranging from 0.9 ppb at Lompoc to 5.5 ppb at Mira Loma. Nitric acid was the next most abundant individual acid, with concentrations ranging from 0.5 ppb at Lompoc to 4.2 ppb at San Dimas. Formic acid was somewhat less abundant, with concentrations ranging from 0.3 ppb at Lompoc to 3.1 ppb at Upland. Hydrochloric acid levels were low everywhere, ranging from 0.3 ppb at Atascadero to 1 ppb at Long Beach. Overall, the gaseous acid data indicate there is approximately a factor of six difference in acid concentrations between the least and most polluted communities. Despite this wide range in acid concentrations it may be difficult to isolate the effect(s) of acids from that of other pollutants because the communities low in acid, and high in acid coincide with those communities having low and high concentrations of other pollutants.

The 1994 nitric acid levels are higher than those measured in 1986 at Long Beach (2.5 versus 1.3 ppb) and Upland (3.9 versus 2.3 ppb) by Solomon et al., (1988). Perhaps the most surprising data were for Mira Loma and Riverside, where 2.6 and 3.2 ppb of nitric acid were measured in 1994. These levels were substantially higher than the 0.7 ppb measured by Solomon et al., (1988) at the Riverside-Rubidoux site, which is located between Mira Loma and Riverside. The differences may be a result of reductions in the ammonia emissions from dairy operations in the areas or due to differences in the measurement technologies employed. The 1994 data indicate Mira Loma and Riverside are moderate nitric acid sites and high total gaseous acid sites.

There is potential bias in $PM_{2.5}$ sulfate, nitric acid, and acetic acid data from the TWS, as described in Lurmann et al., 1994. The sulfate levels may be biased low by up to 10 percent due to electrostatic losses in the inlet. The nitric acid may be biased high by up to 30 percent, perhaps by collection of nitrous acid on the denuder and subsequent conversion by ozone to nitrate. The acetic acid data may also be biased high due to interference from peroxyacetyl nitrate (PAN).

For assessment of the sampler precision, there were one to three collocated samplers at any of the 12 communities during each sampling period. The percent difference in measured concentration (i.e., the difference between the collocated measurements divided by the mean concentration and multiplied by 100) for each pollutant are shown in Table 3-10. Only samples with concentrations above the lower detection limit were included in the calculation. The Lower detection limits (LOD) were determined during the pre-study sampler development (Lurmann et al., 1994) and are also included in Table 3-10. For $PM_{2.5}$ chloride and hydrochloric acid, there were insufficient samples above LOD to calculate the precision.

The original goal during sampler development was to provide a sampler with a precision of ±7.5 percent for all components, which was achieved in several small-scale pre-study trials. However, given the requirements of the sampling protocol (the sampler operates unsupervised for two-week periods and the same sampler is out in the field for

long periods of time), a more realistic goal for precision is about 15 percent, which is sufficient for differentiating the concentrations between the 12 study communities. By this benchmark, the sampler performed adequately for all pollutants having a sufficient number of samples above the LOD in order to calculate the precision.

3.2.3 Comparison of Pollutant Profiles With Design Profiles

One of the most important exposure assessment tasks in Phase II of the study was comparison of the actual pollutant profiles with the design profiles. An evaluation was performed to determine how well the 1994 ambient pollutant profiles for the various communities compared with the 1986 to 1990 design profiles. The design profiles for ozone, NO_2, and PM are presented in Table 3-11 along with estimated profiles for 1993 and the actual profiles for 1994. The design and actual profiles for acids and $PM_{2.5}$ aerosols are shown in Table 3-12. The exposure parameters used for the design were the annual average daily 1-hr maximum ozone, 24-hr NO_2, 24-hr PM_{10} measured by HiVol samplers, and the sum of 24-hr inorganic acid concentrations.

Tables 3-11 and 3-12 show the normalized deviations of the annual concentrations from the mean values in the 12 communities. Tables 3-13 and 3-14 show the ranking of the communities, where communities with deviations less than -0.5σ were ranked as having low pollutant levels, communities with deviations greater than +0.5σ deviations were ranked as having high pollutant levels, and communities with $\pm$0.5σ deviations were ranked as having medium or moderate pollutant levels, as discussed in Section 2.5.

3.2.3.1 Ozone

The 1994 deviations for ozone were greater than expected based on the 1986 to 1990 design data by almost 1σ at Lake Arrowhead and less than expected by about 0.5σ at San Dimas, Upland, and Mira Loma. Lake Arrowhead stands out as an exceptionally important high ozone community for this type of study, and in fact its high levels may bias the 1994 12-community mean significantly upwards.

The ranking of relative ozone levels in the communities is the same in 1994 as 1986 to 1990, with the exception of Mira Loma and Upland which have medium rankings instead of high rankings, based on the annual average data. However, the relative ozone rankings during the high ozone season (May to September, 1994), shown in Table 3-13, are identical to the design rankings in all communities (i.e., Mira Loma and Upland rank high in the extended summer period as expected in the design). The relative rankings based on actual and interpolated 1993 data are also the same as the design rankings. Thus, the actual ozone profiles in the communities are quite similar to the design profiles.

The 12-community mean daily 1-hr maximum ozone level was lower (67 ppb) in 1994 than in 1986 to 1990 (75 ppb); however, it was higher than in 1993 (62 ppb). Over time, we expect ozone levels in the more polluted communities will decrease more than in the less polluted communities, so the between community variations in ozone may be diminished. However, we expect Lake Arrowhead will stand out as a high ozone community for many years to come.

3.2.3.2 Nitrogen dioxide

The 1994 NO_2 levels were higher than expected in Atascadero and Lancaster, and lower than expected in Long Beach and Lake Arrowhead, based on the 1986 to 1990 data employed in the design. The relative rankings of NO_2 levels in the communities was the same in 1994 as 1986 to 1990, with the exception of Lancaster and Lake Arrowhead. Lancaster ranks as having moderate rather than low NO_2 levels based on the 1994 data, which may reflect population growth in the area. Lake Arrowhead ranks as having low NO_2 levels, rather than the medium levels expected from the design. Atascadero still ranks as a low NO_2 community and Long Beach still ranks as a high NO_2 community, based on the 1994 data. The 12-community mean NO_2 levels were 21, 25, and 25 ppb in 1994, 1993, and 1986 to 1990, respectively, which may suggest a modest downward trend in NO_2 or simply reflect differences between measured and interpolated values. Overall, the actual pollutant profiles for NO_2 are quite similar to the design.

3.2.3.3 Particulate matter

Evaluation of the design for PM_{10} is somewhat confounded by the differences in HiVol and TEOM measurement methods. The 12-community mean PM_{10} levels were 53 and 42 $\mu g/m^3$ in 1986 to 1990 and 1993, based on data collected with HiVol samplers. The 12-community mean PM_{10} concentration was 29 $\mu g/m^3$ in 1994, based on data collected with TEOM samplers. The 1994 TEOM PM_{10} levels were lower than expected from the HiVol PM_{10} data in all communities.

Accounting for differences in the measurement techniques, the relative PM_{10} levels were higher than expected in Atascadero, Santa Maria, Long Beach, and Mira Loma, and lower than expected in San Dimas and Riverside. The 1994 relative rankings of community PM_{10} levels were the same as in the design, with the exception of San Dimas and Riverside. San Dimas and Riverside rank as moderate for PM_{10} based on the 1994 data, instead of high. The 34 $\mu g/m^3$ mean PM_{10} concentration in Riverside in 1994 is surprisingly low compared to the 55 $\mu g/m^3$ level observed at the neighboring community of Mira Loma. The other communities met expectation.

Comparison of the 1994 relative rankings of communities for $PM_{2.5}$ with the PM_{10} design shows excellent agreement with the exception of Lancaster. Lancaster has low $PM_{2.5}$ levels in contrast with the moderate PM_{10} levels expected from the design.

San Dimas and Riverside both rank as high based on the $PM_{2.5}$ data. Thus, the relative ranking based on $PM_{2.5}$ data agrees better with the PM design than that based on the TEOM PM_{10} data.

Overall, the 1994 measurements indicate the PM levels in the communities are similar to the design expectations. Ten of 12 communities had the same rankings as in the design based on the PM_{10} data; 11 of 12 communities had the same rankings as the design based on the $PM_{2.5}$ data. All of the differences are minor except those at Riverside.

The concentrations observed at Riverside were unexpected, given the historical measurements in this region (e.g., Chow et al., 1992). Therefore, a special investigation was initiated in Phase III to determine the accuracy of these measurements. For a period of several months in 1995, a collocated TEOM was installed at Riverside to determine if the low concentrations are correct or are the result of problems with the TEOM that is currently installed. These data will be analyzed in Phase III.

3.2.3.4 Gaseous acids

The relative ranking of inorganic acid concentrations is the same as the design ranking in seven communities and different in five communities. Atascadero, Lompoc, and Santa Maria rank as having low inorganic acid levels, as expected. San Dimas and Upland have high inorganic acid levels, as expected, and Lake Elsinore and Lake Arrowhead have moderate inorganic acid levels, as expected. However, Lancaster and Long Beach have low and moderate rather than high inorganic acid levels, based on the 1994 data. Alpine and Mira Loma have moderate rather than low inorganic acid levels, and Riverside has high rather than low inorganic acid levels. Thus, three communities moved from the extreme to the moderate category and two communities, Lancaster and Riverside, moved to the opposite extreme. These differences, especially at Riverside, are larger than expected, however, less inorganic acid data were available for the design than for other pollutants. The absence of low acid levels at Riverside and Mira Loma (where other pollutants are high) may compromise the study's ability to separate the health effects of inorganic acids from those caused by ozone, PM_{10}, and NO_2. The low acid levels at Riverside and Mira Loma were unexpected, given the measurements collected by Solomon et al., 1988.

Organic acid data were not used for the design. The community rankings shown in Table 3-14 indicate the organic and inorganic acid rankings are similar. The only differences are that Lancaster has moderate organic acid levels compared to its low inorganic acid levels, and Mira Loma has high organic acid levels, versus moderate inorganic acid levels.

3.2.4 Summary of Ambient Air Quality in the 12 Communities

The principal findings regarding ambient air quality in the 12 communities are summarized below.

- Ambient ozone levels in 1994 in the communities are similar to the levels anticipated in the Phase I design. Lompoc, Santa Maria, Atascadero, and Long Beach have low ozone levels. San Dimas, Riverside, and especially Lake Arrowhead have high ozone levels. Upland and Mira Loma have high ozone in the May to September high ozone season, but rank as moderate for ozone on an annual basis. Lancaster, Lake Elsinore, and Alpine have moderate ozone levels. The only exceptions from the design classifications were the annual average ozone at Upland and Mira Loma. In addition, Lake Arrowhead stands out as having exceptionally high ozone by all measures of exposure. For example, during May to September the average daily 1-hr maximum ozone is 137 ppb at Lake Arrowhead compared to a 12-community mean of 90 ppb and a low of 35 ppb at Lompoc.

- Ambient NO_2 levels in 1994 in the communities are similar to the anticipated levels. Lompoc, Santa Maria, Atascadero, Alpine, and Lake Arrowhead have low ambient NO_2 levels. Long Beach, San Dimas, Upland, Riverside, and Mira Loma have high NO_2 levels. Lancaster and Lake Elsinore have moderate NO_2 levels. The only exceptions to the design classifications were Lake Arrowhead, which ranked low rather than moderate, and Lancaster, which ranked moderate rather than low.

- Ambient PM_{10} levels in 1994 in the communities are also similar to the anticipated levels. Lompoc, Santa Maria, Atascadero, Lake Arrowhead, and Alpine have low ambient PM_{10} levels. Upland and especially Mira Loma have high PM_{10} levels. Lancaster, San Dimas, Long Beach, Lake Elsinore, and Riverside have moderate PM_{10} levels. The only exceptions to the design classifications were San Dimas and Riverside, which ranked moderate rather than high.

- Ambient $PM_{2.5}$ levels are approximately 65 percent of PM_{10} levels. The ranking of communities based on the 1994 $PM_{2.5}$ data agreed well with the anticipated PM_{10} levels. The only exception to the design classification was at Lancaster, where the 1994 $PM_{2.5}$ levels ranked as low instead of moderate.

- Nitrate, ammonium, sulfate, and chloride accounted for 37 to 61 percent of the $PM_{2.5}$ on a mass basis. Ammonium nitrate accounted for 14 to 47 percent of the $PM_{2.5}$. More than 45 percent of the $PM_{2.5}$ was ammonium nitrate in the Mira Loma and Riverside communities. Ammonium sulfate accounted for 11 to 26 percent of the $PM_{2.5}$.

- The ambient concentrations of nitric acid and hydrochloric acid are only moderately similar to the anticipated levels. Atascadero, Lompoc, and Santa Maria have low inorganic acid levels, as expected. San Dimas and Upland have high inorganic acid levels, and Lake Elsinore and Lake Arrowhead have moderate inorganic acid levels, as expected. However, Lancaster and Long Beach have moderate rather than high inorganic acid levels, and Alpine and Riverside have moderate rather than low inorganic acid levels. Most surprising, Mira Loma has high rather than low inorganic acid levels. The differences, especially at Mira Loma, are larger than expected, however, less inorganic acid data were available for the design than for other pollutants. The absence of low acid levels at Riverside and Mira Loma (where other pollutants are high) may compromise the study's ability to separate the health effects of inorganic acids from those caused by ozone, PM_{10}, and NO_2. (Please see section 4.8 for further discussion).

- Ambient concentrations of organic acids (formic + acetic acid) generally exceed the inorganic acid levels. Acetic acid is the most abundant acid in all of the communities. Formic acid is less abundant than nitric acid. Hydrochloric acid is the least abundant acid and is barely measurable at most sites.

- Overall, 1994 air quality ranked exactly as expected for all four pollutants in 4 of the 12 communities (Atascadero, Santa Maria, Lompoc, and Lake Elsinore). 1994 air quality in 5 of the 12 communities differed from the design ranking for one of the four pollutants (in San Dimas, Upland, Long Beach, Lake Arrowhead, and Alpine). The 1994 pollutant profiles in Mira Loma, Riverside, and Lancaster indicated two pollutants matched the design expectations and two other pollutants did not. In almost all cases, the deviations from expected ranking were between low and moderate, or moderate and high, rather than low and high. The actual air quality conditions in the communities ranked moderate (rather than high or low) more often than expected from the design analysis.

3.3 PRE-STUDY LIFETIME AMBIENT AIR POLLUTION EXPOSURE ESTIMATES

In order to examine the relationships between health status and cumulative exposure to air pollution in the cross-sectional study, estimates were needed of the population's exposure to all pollutants under study from birth to the time at which the first health status questionnaire and first pulmonary function tests (PFTs) were

administered. The first questionnaire and PFTs were collected in early 1993. Because air pollutant concentrations vary widely with location and time, a goal was established to estimate concentration on a monthly basis in all of the locations where the children lived. The time frame of interest was from 1976 to 1992 for the oldest children in the study. Compilation of the exposure estimates on a monthly basis allows the examination of health effects for various periods in the children's lives, such as the first 5 years or the most recent 5-year period.

Residential history questionnaires were collected in 1993 to provide the approximate locations of the students' residences on a monthly basis from birth through 1992. The first year's residential history data indicated that in addition to the 12 cities in which the children lived in 1993, the children had lived in 453 other California cities or rural areas, 126 non-California U.S. cities or counties, and 121 cities in 36 foreign countries. The majority of children had always lived in California. However, a modest number of children spent significant periods of time in other states and in foreign countries. The California, non-California U.S., and foreign locations specified in the residential history questionnaires are listed in Tables 3-15, 3-16, and 3-17, respectively.

The only data available to characterize pre-study exposures of the population are ambient air quality data. These data are not available for all of the pollutants, locations, and/or time periods of interest. No reliable pre-study housing information or time-activity data exist for these children, so the use of a microenvironmental approach to characterize pre-study exposures is not possible. The accuracy of historical exposure estimates based on ambient data alone may be poor because of the many factors known to influence personal exposure. However, as a practical matter, given the choice between using historical ambient data or no exposure data at all for the cross-sectional study, it was relatively easy to accept using the ambient data. The inherent limitations of using ambient data alone need to be recognized.

3.3.1 Ambient Air Quality at California Residences

Ambient air quality concentrations for ozone and NO_2 on an hourly basis, and TSP and/or PM_{10} concentrations on a 24-hr basis (every sixth day) were acquired from ARB for all California locations with measurements between 1976 and 1992. Measurements were made at over 700 locations in California during this period. A core set of about 100 stations collected data throughout this period. Thus a rather large database was available for California. These data tended to be quite complete. The ARB indicated that the archived pre-1980 NO_2 data had a 14-percent positive bias; these data were adjusted downward prior to use in this study.

A number of characteristics of the database are worth noting. The PM data were TSP, rather than PM_{10}, before 1986. The PM_{10}/TSP ratios from 78 stations were analyzed with concurrent data in the 1986 to 1992 time period. The ratios varied from 0.27 to 0.78, and had a mean of 0.55. The historical TSP data were converted into

estimated PM_{10} using the mean ratio of 0.55. Too few stations had both PM_{10} and TSP to consider using site-specified conversion factors. Unfortunately no ambient data for acidic species were available on a statewide or long-term basis. Hence exposure to acids was not considered in the analysis. The locations of approximately 50 of the 700 stations are uncertain. The stations with missing locations were typically ones from short duration measurement programs that were not particularly useful for this study's purposes. Nevertheless, an attempt was made to estimate the location of all of the historical measurement sites from the names and county codes of the sites.

A number of ozone exposure metrics, in addition to the 24-hr average NO_2 and PM_{10}, were generated from the data to satisfy the needs of health researchers. The parameters included in the analysis were:

- Average (24-hr) ozone
- Average 10:00 a.m. to 6:00 p.m. ozone
- Average 1-hr daily maximum ozone
- Number of hours with ozone greater than 60 ppb
- Number of hours with ozone greater than 90 ppb
- Number of hours with ozone greater than 120 ppb
- Number of hours with ozone greater than 150 ppb
- Number of concentration-weighted hours (ppb-hr) with ozone greater than 60 ppb
- Number of concentration-weighted hours (ppb-hr) with ozone greater than 90 ppb
- Number of concentration-weighted hours (ppb-hr) with ozone greater than 120 ppb
- Number of concentration-weighted hours (ppb-hr) with ozone greater than 150 ppb
- Average (24-hr) NO_2
- Average (24-hr) PM_{10} (or estimated PM_{10})

Monthly values of these parameters from 1976 through 1992 were generated for all stations with more than 75 percent valid data.

To have the most geographically representative data available to construct lifetime exposure estimates, geographic coordinates were assigned to all locations specified in the students' residential history database, and the monthly ambient air quality data were interpolated to these locations. Generally the UTM coordinates of the centroid of the municipality reported by the student was used as the location. About 50 of the 450 California locations were rural areas, and coordinates were assigned that were believed to be at least within 50 km of the actual locations. A simple spatial interpolation scheme was employed. If a station with valid data for a given month was located within 5 km of a residence location, the data from that station were assigned to the residence location. If no station with valid data was located within 5 km, inverse

distance squared weighing of the data from the three stations closest to the residence location was used, provided three stations (with valid data) were within 100 km of the residence location. Data from fewer stations were used in the interpolation in data-sparse regions where fewer than three stations within 100 km existed or had valid data. The 100-km limit is rather large but was needed to permit assignment in rural areas. These interpolated historical ambient air quality data were incorporated into the main project database at USC. The lifetime exposures of individuals who had always lived in California were calculated by matching these data with the residential history data.

3.3.2 Ambient Air Quality in Other Areas

The assignment of historical exposures for students who lived in foreign countries and in states other than California required a somewhat different approach because of the lack of ambient air quality data and lack of specificity in the geographic locations of residences. Historical air quality data for hourly ozone and NO_2, and once every sixth day 24-hr TSP and PM_{10} collected from 1976 to 1992, were acquired from the U.S. EPA's Aerometric Information Retrieval Service (AIRS) database for the counties outside California in which the students lived (i.e., for the areas listed in Table 3-16). The amount of data available for these pollutants and cities generally falls short of what is available for California in terms of the station density and length of measurement record.

Nonetheless, monthly averages were calculated for each pollutant using the linear interpolation algorithm described above to calculate monthly averages for ozone, nitrogen dioxide, and PM_{10}. To have a valid monthly average, 75 percent of the data needed to be available. For periods prior to the beginning of PM_{10} measurements, the TSP concentrations were used to estimate PM_{10} as described for the California data. The application of the spatial interpolation was limited by only having data for the counties that the children lived in, and not necessarily the adjacent counties, which, particularly in the East, could be used in the interpolation. In the future, all data within the maximum interpolation range will be acquired and used. Generally, there were significant gaps within the interpolated exposures, and, for many counties, there were no data available. However, there is a broad distribution of ozone concentrations across the country, and this technique should predict differences in the lifetime exposures between children who have lived in areas of the country with significantly different concentrations. These predictions should improve when all United States data are included in the interpolation (in Phase III).

Comparable ambient air quality data are not readily available for foreign locations. Air quality summaries from WHO (1992) are available for 9 of the 118 foreign locations: Bangkok, Bombay, Buenos Aires, Jakarta, London, Manila, Mexico City, Seoul, and Tokyo. These summaries are used for approximate classification of the cities as clean, moderate, or dirty with regard to the pollutants of interest. Classification of the other foreign locations was made based on the approximate population of the

cities. Rural locations were assigned to the clean category, while urban locations were mostly assigned to the moderate category. The largest urban areas were assigned to the most polluted category. Clearly these assignments are uncertain, and careful consideration must be made regarding use of these assignments in the epidemiologic analyses. The lifetime exposure estimates for students who had lived in foreign countries for only a short period of time may be reasonably accurate, whereas the lifetime exposure estimates for students who lived most of their lives in foreign countries are highly uncertain.

3.4 INDOOR AND OUTDOOR OZONE AT SCHOOLS

3.4.1 Measurement Program Design

Characterization of ozone concentrations indoors and outdoors at schools was given a high priority for Phase II because little was known about ozone at schools at the start of this program and because children spend a significant amount of time at schools. The objectives of the school ozone monitoring program were (1) to collect sufficient indoor and outdoor ozone concentration data at schools to classify all schools with respect to their indoor/outdoor ozone relationships, (2) characterize the variability of the ozone indoor and outdoor concentrations between schools, and (3) to collect outdoor ozone concentration data at schools to characterize the within-community variation in ozone air quality. Since there are 50 schools in the study and since many have a mixture of building types and HVAC system types, only a subset of the children's classrooms could be monitored. In fact, in planning the study, it was determined that it would be prohibitively expensive to use continuous hourly monitors in all schools. An alternate sampling approach was developed that involved using a combination of continuous hourly instruments in a few classrooms and inexpensive portable TED samplers in two classrooms in all schools and outdoors.

A quiet, portable, battery-operated TED ozone sampler was developed specifically for classroom sampling for this study. The sampler provides controlled airflow across two single-ended Ogawa badges with sodium nitrite-coated filters for measuring ozone by the Koutrakis method (Koutrakis et al., 1993). The sampler has a 7-day timer so that it can be set to sample for specific hours of the day on specific days of the week. This capability was needed for the school sampling since the only ozone concentrations of interest were those measured during the conventional school day. Laboratory evaluation of the sampler showed that it was not affected by interference from NO_2, nitrous acid, PAN, or SO_2. However, it does respond to nitric acid and hydrogen peroxide, both of which generally have low ambient and indoor concentrations relative to ozone. Field evaluations of the TED ozone sampler showed that it measures ozone with a +6 percent bias and ±12 percent precision on average compared to a continuous monitor under ambient conditions. The development and evaluation of the TED sampler are documented in Lurmann et al., 1994.

The school TED sampling program consisted of measuring ozone concentrations inside two classrooms and at one outdoor location, usually on the roof, of each school for 2 weeks in Fall 1993 and 2 weeks of Spring 1994. During each week each TED sampler collected a 4-day integrated ozone sample for the hours children attended schools (either 8:00 a.m. to 3:00 p.m., or 7:00 a.m. to 2:00 p.m., depending on the school). The four integrated sampling periods were considered the minimum needed to classify a school (or classroom) with regard to its indoor/outdoor ozone relationship and to assess how outdoor ozone at the school compared to the ozone levels at the community monitoring station. The criteria for selecting classrooms were to use only those rooms that study participants occupied for at least 1 hour a day (i.e., because high school students change classrooms every hour) and rooms that represented a range of building and ventilation types. Strict criteria were developed for the placement of samplers within the classrooms and on the roofs to minimize potential influence from nearby vents, windows, doors, or walls. The samplers were placed on 3-ft stands, usually more than 2 ft away from the walls in the classrooms and 30 ft or more away from trees or rooftop vents outdoors. In addition, "classroom site surveys" and "school site surveys" were compiled to document the site characteristics, including the sampler placement within the room or on the roof; the building type (conventional or portable); the type of heating, ventilating, and air conditioning (HVAC) system; the use of windows; and the type of floor, wall, and ceiling materials. The field operating procedures and data processing for the school TED sampling are described in Lurmann et al., 1994.

Limited continuous monitoring of hourly ozone was conducted in selected classrooms. The objectives of the continuous monitoring were to assess the validity of the integrated ozone obtained using the TED sampler, and to investigate the extent of short-term variations in indoor/outdoor ratios, and the extent of week-to-week and seasonal variations in the indoor/outdoor ratios. To address these measurement objectives, continuous ozone monitors were installed and operated in one classroom for 7 months and in four other classrooms for 4 to 5 weeks. The criteria for selecting communities and classrooms were to include both high- and low-ozone communities and to select classrooms with different building types and HVAC system types. The field operating procedures and data processing for the continuous monitoring in schools are described in Lurmann et al., 1994.

3.4.2 Results of TED Ozone Sampling

The school participation rate in the TED ozone sampling was high; 48 of the 50 schools with children at the beginning of Phase II permitted sampling. The exceptions were one elementary school in Santa Maria that denied permission because of a prior unsubstantiated episode of "sick-building syndrome" in the school and one middle school in Long Beach that was having major remodeling work done. Because of a limitation on the number of TED samplers and technicians, it was not possible to sample all schools at the same time. The schools were divided into four groups for

sampling in the August to October 1993 period (Fall) and four slightly different groups for the April to June, 1994 sampling period (Spring). The grouping of schools in the Fall was based on a desire to complete the sampling before the end of the "ozone season," which was interpreted as the end of October. Thus the school opening schedules largely determined the groupings in the Fall. Whenever possible, schools within a community were sampled concurrently. In fact, in the Spring, all schools in each community were sampled concurrently, which provided optimum data to characterize within community outdoor ozone variability. Also, because ambient ozone concentrations generally decline over the Fall sampling period and increase over the Spring sampling period, the approximate order in which communities were sampled was maintained so that schools that were sampled in the lower ozone portion of the Fall were sampled in the higher ozone portion the following Spring. The schedules for the Fall and Spring sampling (groups 1 to 8) are shown in Figures 3-37 and 3-38.

Many of the schools in the study have a mixture of building types, floor, wall, and ceiling materials, and HVAC systems. The classrooms selection for sampling can be categorized into six groups:

1. Conventional buildings with carpet and air conditioning
2. Conventional buildings with carpet, without air conditioning
3. Conventional buildings without carpet, with air conditioning
4. Conventional buildings without carpet or air conditioning
5. Conventional buildings with swamp coolers
6. Portable buildings with carpet and air conditioning

The majority of schools had both carpet, which is likely to readily absorb ozone, and air conditioning, which if used, tends to diminish indoor ozone. Most schools had windows that could be opened; however, few, if any, of the classrooms had good cross ventilation (e.g., an open door across from open windows) that would promote high air exchange rates. Two different types of classrooms were selected for sampling at most schools; often, one classroom was in a conventional building, and the other classroom was in a portable building. At the schools, the outdoor sampling was performed at a rooftop location near the classrooms being monitored within a school. The rooftop locations were used to ensure the security of the sampler, although one outdoor sampler was vandalized in the Fall.

The Koutrakis method is a filter-based measurement method that requires deployment of filter blanks. Both trip blanks and field blanks were used in the program. The trip blanks were taken to the sampling location and opened for short time intervals upon filter installation and filter removal, and kept in sealed containers within the TED sampler at all other times. The field blanks were opened and installed in a non-operating TED sampler for 1 week at a time, but the sampler was not operated. One trip blank was used for every third sampler. One field blank was used in each community. The blanks were equally distributed between indoor and outdoor sampling

locations. The indoor trip blanks, indoor and outdoor field blanks, and outdoor field blanks showed comparable levels in the Fall sampling program, with the outdoor field blanks showing slightly higher levels of ozone. The mean of the trip blanks was 0.25 mg/l nitrate for the Fall and 0.10 mg/l nitrate for the Spring (see Lurmann et al., 1994 for the conversion between mg/l and ppb for typical sampling intervals). These values were subtracted from the indoor nitrate values before calculating indoor concentrations of ozone. Similarly, the mean of the outdoor field blanks (0.33 mg/l for the Fall and 0.18 mg/l for the Spring) was used to calculate outdoor concentrations. The trip blank variability in the Fall was comparable to that observed during the sampler development. Based on the blank variability data, the lower detection limit [LOD = 3 times the standard deviation of the blanks] of the TED sampler for the 28-hr sampling interval was 8 ppb for the Fall and 11 ppb for the Spring.

Collocated TED sampling was performed in every community for quality assurance. Figures 3-39 and 3-40 show a comparison of collocated ozone data collected indoors and outdoors during the Fall and the Spring, respectively. The correlation coefficient, r^2, between the paired samples is 0.87 for the Fall and Spring data combined. The correlations were similar for the Fall and Spring data separately. This shows the sampler is able to provide reasonably precise data for the purposes of the study. During the final Spring sampling period, the batch of passive monitors received from Harvard had very high and variable blank levels, indicating that there was some contamination during preparation or transport. These high blank levels led to many non-physical (i.e., negative) concentrations. Thus, the Period 8 data was not used. Period 8 was expected to be when the highest ozone concentrations for the Spring sampling phase were observed. Thus, the loss of this data results in lower concentrations for the Spring than the Fall. Additionally, fires near Buena Vista Elementary School in Lompoc during Spring sampling resulted in contaminated nitrate filter samples and unrealistically high ozone levels.

3.4.2.1 Indoor and outdoor ozone concentrations and ratios

Scatter plots of the indoor and outdoor integrated ozone concentrations are shown in Figures 3-41 and 3-42 for the Fall and Spring respectively. The indoor concentrations ranged from below the detection limit to 40 ppb, while the outdoor concentrations ranged from 10 to 120 ppb. Almost half of the indoor ozone concentrations were below the detection limit for the Fall and more than half were below the detection limit for the Spring, which was a somewhat surprising result given that the outdoor concentrations were moderate during both periods. Some of the indoor concentrations were so low that filters had less nitrate than the mean blank level, which produced "unphysical" negative concentrations. A more surprising result, however, was that the maximum indoor concentration was only 40 ppb, which is comparable to background ozone levels in clean remote areas. The outdoor concentrations were typical for this season for the 8:00 a.m. to 3:00 p.m. sampling period. When analyzed as a single group, there is little correlation between the indoor and outdoor ozone, $r^2 = 0.075$.

The correlation in the Spring was slightly higher, likely a result of the higher concentrations observed during this sampling phase. The data are still quite scattered when distinguished by sampling group, as shown in Figures 3-43 and 3-44.

The indoor/outdoor ozone concentration ratios from the Fall and Spring sampling are shown in Figures 3-45 and 3-46. The indoor/outdoor ratios ranged from 0 to 1.3. Only two samples had ratios greater than 1, and these occurred when the outdoor concentration was extremely low (10 to 20 ppb). Ratios above 1 are unphysical because there are no known indoor sources of ozone in schools and we eliminated these samples from subsequent calculations. Because so many indoor concentrations were below the detection limit (LOD) , it is difficult to report an accurate overall mean ratio for the schools. The overall mean indoor/outdoor ratio for the subset of samples with concentrations above LOD was 0.32 with a standard deviation of 0.15. If the below LOD samples are included and ratios are calculated assuming the below LOD concentrations are equal to the LOD, the overall mean indoor/outdoor ratio is 0.31 with a standard deviation of 0.16 which is in good agreement with the previous ratio. If the below LOD samples are included and ratios are calculated assuming the below LOD concentrations are equal to zero, the overall mean indoor/outdoor ratio is 0.12 with a standard deviation of 0.18, which is substantially lower. These calculations bound the mean ratios for these schools and clearly indicate the ratios are low on average. The data also indicate there is a relationship between ratio and the outdoor concentration: the ratios decline as the outdoor ozone increases. This is evident in both the Spring and Fall data. Figure 3-47 shows a histogram of the indoor/outdoor ratio during the Fall and Spring programs. For both periods, most of the values fall between 0 and 0.35. Also, the figures shows that the distribution of indoor/outdoor between the Spring and Fall are very similar.

The week-to-week variability in the indoor/outdoor ozone ratio at schools in several selected communities are shown in Figures 3-48 through 3-52. The plots show there are cases where indoor/outdoor ratios within a classroom agree fairly well among the four sampling periods, but there are also cases where the ratios are significantly different. Table 3-18 shows the mean of the indoor/outdoor ratios stratified by air conditioning type (i.e., swamp cooler or traditional AC), air conditioner use (was it functioning during sampling), building type (conventional or portable), and flooring type (with or without carpet). The data show that the key factor influencing the indoor/outdoor ratio was whether the air conditioner or swamp cooler was being used, with the indoor/outdoor ratio increasing when the air conditioner was off and when the swamp cooler was being used. When the air conditioner is off, the doors to the room are likely to be open, and vice versa. Open doors and windows increase the room air exchange rate, which decreases the residence time of the ozone, thus decreasing its deposition rate. Normally, the swamp coolers were operated while the doors or windows were open, so those results are also consistent with this explanation. The results also indicate that the indoor/outdoor ratios are lower in the portable rooms compared to the conventional buildings. This may be because the portables are not as

leaky as conventional buildings. Thus, they have a longer ozone residence time and a higher ozone deposition. The surprising result is that the presence of carpet, which is expected to readily absorb ozone, does not affect the indoor/outdoor ratio. Thus, it appears that the air exchange rate is more important than the surface composition of a room in determining a room's indoor/outdoor ratio.

The average indoor/outdoor ratios in specific classrooms in the Fall and Spring are compared in Figure 3-53. There is a lot of scatter in the data, yet the Fall and Spring ratios are statistically correlated with r^2=0.33. This shows that there must be some characteristics of particular rooms that influence the indoor concentrations. To further examine if there were differences among individual rooms that resulted in differences in indoor/outdoor ratios throughout the study, analysis of variance (ANOVA) was done. The hypothesis that the indoor/outdoor ratio among all of the rooms was not statistically differentiable was first tested. This test yielded an F-Ratio of 4.21 ($P<0.001$), which is highly significant. This test was also conducted for only points with indoor concentrations above 10 ppb (a nominal detection limit), which yielded an F-Ratio of 1.57 ($P=0.051$), which is marginally significant. These tests suggest that there were differences among the rooms that were evident throughout the study. As shown above, air conditioning is one factor that could result in these differences. To test if there were other factors, the tests were repeated with data stratified by whether the air conditioner was on. With the "air conditioner on" the F-Ratio was 2.68 ($P=0.001$), and with the "air conditioner off" the F-Ratio was 0.60 ($P=0.83$). This shows that within the "air conditioner on" group there are other factors that affect indoor/outdoor concentrations, such as the building type (i.e., conventional or portable building). Within the "air conditioner off" group, no differentiation can be made between the rooms.

3.4.2.2 Within-community variability in ambient ozone concentrations

The school sampling program was designed to facilitate analysis of within-community variability in ambient ozone, since ozone was concurrently sampled at up to five locations. Figures 3-54 through 3-57 compare ambient ozone concentrations collected at different schools in nine communities. They show that in most cases the 8:00 a.m. to 3:00 p.m. integrated ozone concentrations outdoors collected at different schools agreed within ±10 ppb. However, there are cases where the outdoor ozone at different schools within the same community differed by up to 40 ppb [e.g., San Dimas (week 5), Upland (week 11) and Lancaster (weeks 11 and 12)].

Figures 3-58 and 3-59 show a comparison of the 8:00 a.m. to 3:00 p.m. integrated outdoor ozone concentrations measured by the community monitors and the outdoor TED samplers. With a few exceptions, the ozone levels at the schools are within ±20 ppb of the values recorded at the community monitors. For the Fall, ozone levels at the schools exceed those at the community monitoring site in 40 percent of the cases; 60 percent of cases show values at schools below those at the community

monitoring site. For the Spring, ozone levels at the community monitors exceeded the school ozone levels about the half of the time. The six-percent positive bias expected in the TED sampler data (see Appendix A) is not evident in these data. The lack of significant bias in the distribution is a welcome result, because it was anticipated that significantly lower ozone at schools would be found since they are often located near busy roadways where NO scavenges ozone. The roadway effect is clearly not dominant in the data.

3.4.2.3 Ozone concentrations during and after school

The children typically spend ½ to 2 hr outdoors during the normal school day. However, the children may commonly spend most of the hours after school outdoors. Figures 3-60 and 3-61 shows a comparison of the integrated ozone concentrations at the community monitors during the Fall sampling period for 8:00 a.m. to 3:00 p.m. and 3:00 p.m. to 6:00 p.m. The scatter plot shows outdoor ozone levels after the school hours were comparable to those during the school hours and, in some cases, higher than during the school hours. Thus exposures after school could be at least as important or more important than those during school.

Figures 3-62 through 3-67 show selected comparisons of the ozone in classrooms during school hours, outdoor ozone during school hours, outdoor ozone at the community monitor during school hours, and outdoor ozone at the community monitor between 3:00 p.m. and 6:00 p.m. In these examples the outdoor ozone concentrations after school are comparable or higher than those during school hours. In addition, they show that the indoor ozone levels are low and usually insignificant relative to the outdoor levels during and after school. As these plots illustrate, an important finding from this study is that the outdoor exposures to ozone are likely to determine a child's overall exposure to ozone. That is, even though the children spend more hours indoors than outdoors during a typical school day, the cumulative exposure indoors will usually be less than the cumulative exposure outdoors. Thus knowing the amount of time children are outdoors and which hours of the day they are outdoors is critically important in determining the children's exposure to ozone. Fortunately information on the time outdoors by hour of the day is collected in the annual time-activity survey and is used to estimate children's exposure. While this type of survey information has not been used for epidemiologic purposes in the past, we expect it will enhance the accuracy of exposure assignments for individual children.

3.4.3 Continuous Ozone Monitoring Results

Limited continuous monitoring of hourly ozone was conducted in five classrooms during Phase II. The purpose of the continuous monitoring was to assess the validity of the integrated ozone obtained using the TED sampler, to investigate the extent of short-term variations in indoor/outdoor ratios, and to evaluate the extent of week-to-week and seasonal variations in the indoor/outdoor ratios. Continuous ozone

monitors were installed and operated in one classroom for 7 months and four other classrooms for 4 to 5 weeks to address these measurement objectives. The criteria for selecting communities and classrooms were to include both high- and low-ozone communities and to select classrooms with different building types and HVAC system types. As a matter of convenience, schools at which on-campus community monitors were installed for outdoor ozone monitoring were selected. The indoor ozone monitor inlets were located about 6 ft above the floor in the middle of the classrooms in order to minimize the effects of walls, doors, windows, and vents. The classrooms, building characteristics, and sampling periods are described in Table 3-19.

Sampling was initiated October 20, 1993 in the high-ozone community of Mira Loma in a portable classroom at Jurupa Valley High School. The monitor was operated for 7 months with excellent data recovery. The classroom was conveniently located next to the Mira Loma community air monitoring station, which was established at the school for this project. Like all the portable classrooms, it was air conditioned and had both carpeted floors and fabric-covered walls. During the Winter months virtually no ozone was observed indoors. During April and May, 1994, measurable but low ozone was observed indoors. Figure 3-68 shows a plot of the hourly ozone concentrations indoors and outdoors on April 17, 1994, when the outdoor ozone reached a maximum of 140 ppb (which is not unusual for Los Angeles in April). The indoor ozone concentration reached a maximum of 6 ppb at the same hour as the outdoor peak. These data are typical of the results for this classroom. The maximum indoor concentration observed in 7 months of monitoring was about 10 ppb. The average indoor/outdoor ratio was less than 0.05. There were no short-term fluctuations where the indoor/outdoor ratio exceeded 0.20. Since ambient ozone and temperature are highly correlated, the air conditioning system virtually always operated when the outdoor ozone exceeded 80 or 100 ppb. The results from this classroom were in agreement with the TED ozone sampler results from other portable classrooms sampled in the Fall 1993 program.

Based on the results from Mira Loma and the Fall TED ozone sampling, it was decided to monitor four non-air conditioned classrooms in the late Spring to capture the highest ozone before the schools were closed for Summer. Sampling was performed from mid-May to mid-June in two classrooms at Rim of the World High School in Lake Arrowhead, which is the community with the highest ozone in the study, and in two classrooms in Cabrillo High School in Lompoc, which is one of the lowest ozone communities. One carpeted and one non-carpeted classroom were selected at each school. The preliminary results show higher concentrations and indoor/outdoor ratios than observed in the Mira Loma portable classroom. The highest concentrations were observed in Room AS at Rim of the World High School, which has a high ceiling (16 ft), no carpet, and hard surface walls (painted on three sides and mostly glass on the fourth side). The diurnal plot of indoor and outdoor concentrations on the day with the highest indoor concentration (for Room AS) is shown in Figure 3-69. The highest indoor concentration was 70 ppb at 12:00 noon., and the corresponding indoor/outdoor

ratio was 0.50. The indoor ozone decreased dramatically when the classroom was closed and the ventilation system was turned off. The classrooms are usually closed before the time of the peak in the ambient ozone at this school, which again points out the importance of characterizing the students' ozone exposure after school hours.

Diurnal plots of the indoor concentration and the indoor/outdoor ratio for the school operating hours of 7:00 a.m. to 2:00 p.m are shown in Figures 3-70 through 3-73 for both the Lake Arrowhead indoor monitors. The diurnal plots show that the mean indoor ozone concentration is relatively constant throughout the day, but the highest values occur in the middle of the day. Also, the indoor/outdoor ratio is generally higher in the morning and decreases throughout the day. This is unexpected because the air exchange rate should be higher during the warmer part of the day when doors and windows are more likely to be open. However, questionnaire data indicated the teachers often keep the classroom doors closed to minimize noise from the hallway. The diurnal plots show that Room DX had both a higher indoor ozone concentration and indoor/outdoor ozone ratio. Room DX was carpeted, while Room DW had painted walls. Therefore, the deposition effort hypothesized for carpet was not observed, which was consistent with results of the TED sampling.

3.4.4 Summary of Ozone at Schools

At the start of Phase II little was known regarding the ozone levels at schools in Southern California. From our analysis of the school ozone data, a number of accomplishments and findings are notable.

- A TED ozone sampler that uses the Koutrakis ozone measurement method was developed and evaluated to facilitate economical and concurrent sampling in a large number of schools. Laboratory evaluation of the sampler showed that it was not affected by interference from NO_2, nitrous acid, PAN, or SO_2. However, it does respond to nitric acid and hydrogen peroxide, both of which generally have low ambient and indoor concentrations relative to ozone. Field evaluations showed the device is able to measure ozone with a +6 percent bias and ±12 percent precision on average.

- Ozone data were collected in two classrooms and one rooftop outdoor location in 48 of the 50 schools with children participating in the study in Fall 1993 and Spring 1994. The schools included a variety of building types, floor, wall, and ceiling materials, and HVAC systems. The majority of the schools had carpeted floors and air conditioners (see Table 3-18).

- Indoor/outdoor ratios were lowest when the air conditioners were operating, indicating that the air exchange rate is the key variable affecting indoor/outdoor ratios. The presence of carpet did not influence the indoor/outdoor ratios based on the TED sampling data.

- Integrated ozone data from the Fall TED sampler for 8:00 a.m. to 3:00 p.m. indicate indoor ozone levels were low relative to outdoor levels in almost all schools. The indoor concentrations ranged from below the detection limit to 40 ppb, while the outdoor concentrations ranged from 10 to 120 ppb. Almost half the indoor ozone concentrations were below the detection limit of 8 ppb, which was a somewhat surprising result. The most surprising result was, however, that the maximum indoor concentration during this period was only 40 ppb, which is comparable to background ozone levels in clean remote areas.

- Indoor/outdoor ratios ranged from 0 to 1.3. High indoor/outdoor ratios were associated with low outdoor concentrations. The mean indoor/outdoor ratio from cases where the indoor concentration exceeded the LOD was 0.32. However, there were many samples with indoor concentrations below the LOD, so the actual mean is probably lower. the data also indicate there is a relationship between the ratio and the outdoor concentration: the ratios decline as the outdoor ozone increases. The low indoor ozone levels and low indoor/outdoor ratios are believed to be due to the prevalence of carpeted surfaces and air conditioning in the schools.

- School outdoor ozone data indicate there was modest within-community variance in ambient ozone concentrations. In most cases the 8:00 a.m. to 3:00 p.m. integrated ozone concentrations outdoors collected at different schools was within ±20 ppb of the levels recorded at the community monitoring station. In other cases differences as large as 40 ppb (80 versus 120 ppb) were observed.

- Some children may spend more time outdoors after school hours than during school hours. Outdoor ozone levels after school hours (3:00 p.m. to 6:00 p.m.) were comparable to those during school hours (8:00 a.m. to 3:00 p.m.). In some cases, such as in Lake Arrowhead, ozone was usually higher after school than during school hours. Thus exposures after school hours could be as important or more important than exposures during school hours.

- Since indoor ozone levels were low and probably insignificant relative to outdoor levels during and after school hours, outdoor exposures are likely to determine a child's overall exposure to ozone. Even though children may spend more hours indoors than outdoors during a typical school day, the cumulative exposure indoors will usually be less than the cumulative exposure outdoors. Thus knowing the amount of time children are outdoors and which hours of the day they are outdoors are critically important in determining the children's exposure to ozone. Information on the time outdoors by hour of the day is collected in the annual time-activity survey employed in this project, and the data are being used to estimate children's exposure.